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of

HON. JACK DAVIS, M.P.

Minister of Fisheries and Forestry

A Comparison of Aquatic Communities in the Bow River Above and Below Sources of Domestic and Industrial Wastes from the City of Calgary

by R. H. Kussat

Department of Fisheries of Canada, Vancouver, B.C.

Abstract

A general biological survey was conducted on two backwaters of the Bow River, one above, the other below the heavier outfall of industrial and domestic wastes from the city of Calgary. Analyses of biological and chemical data indicated the following changes in the environment had taken place: reductions in numbers of invertebrate genera comprising the bottom fauna, increased coefficients of condition of *Catostomus commersoni* and of *C. catostomus* taken at the downstream locality, and differences in dissolved oxygen, nitrogen and phosphate concentrations. Further, preliminary parasitological surveys indicating reduced incidence and number of certain internal parasites suggest that the intermediate hosts were unable to survive in the more heavily polluted reach of the Bow River. The results of this survey strongly suggest that industrial and domestic effluents discharged into the river are responsible for the differences observed in the study sites upstream and downstream from Calgary.

Introduction

A virginal stream community contains numbers of species in ecological equilibrium. However, with the introduction of industrial and domestic wastes many of the indigenous groups disappear and only groups of organisms tolerant to these effluents survive and often increase in numbers.

Several methods have been established to ascertain whether or not the changes caused by different effluents are harmful to a stream. Chemical and physical water analyses, which until recently have been the usual processes, have told us little of the actual effects of the effluents on aquatic life. Therefore, greater emphasis is now being placed on biological surveys which, when carried out in conjunction with chemical and physical analyses of water, give much more reliable measures of prevailing stream conditions. Recent studies emphasizing the importance and reliability of biological investigations were carried out by Patrick (1949), Hynes (1960), Beak (1964), and Carr and Hiltunen (1965).

Because waste materials from oil refineries and various other industries and from partially treated domestic sewage are piped into the Bow River at Calgary, Alberta, this investigation was initiated to measure some of the environmental factors and

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the effects which the above-mentioned effluents impose on the organisms in areas downstream from Calgary (see map). The Bow River originating in the Canadian Rocky Mountains at Lakes Bow and Hector follows an easterly course to the city of Calgary from where it continues in a southeasterly direction until it joins the Oldman River. Several hydroelectric power developments upstream from Calgary control the water-flow and keep it fairly constant except during spring run-off.

Sampling was carried out in two backwashes of approximately equal size (100 meters long, 50 meters wide and 1-2 meters deep) located within the city limits one above and the other below the heavy outfall of industrial and domestic wastes from the city. These areas were chosen because they afforded an environment relatively undisturbed by seasonal fluctuations of water levels, and ready access for biological sampling. The river bed in the backwashes used for this investigation is generally rocky with the bottom covered with mud and organic debris.

Investigations carried out on the Bow River in the vicinity of Calgary prior to the present study are limited. The Alberta Department of Public Health has checked the chemical properties of the water below the city from time to time (Rolston, 1962; Ferguson, 1965). During the summer months of 1963 and 1964, a study conducted by the Alberta Department of Lands and Forests, Fish and Wildlife Division, on the biological life in the river indicated that qualitative as well as quantitative differences in biological organisms exist between areas immediately upstream and downstream from Calgary (Kussat, 1966).

FIGURE 1. *Upstream backwash in early May, 1965.*

FIGURE 2. *Downstream backwash in early May, 1965.*

The study consisted of four parts: (1) a survey of the organisms constituting the bottom fauna of these backwashes (Figures 1 and 2); (2) a comparison of length, weight, age and stomach content analyses of populations of Catostomidae found in the two areas; (3) a determination of water quality consisting of physical and chemical measurements; (4) a preliminary study of the distributional pattern of some intestinal parasites recovered from the alimentary tract of the three species of catostomids.

Materials and Methods

Each backwash was sampled approximately every twenty days. Nine collections were made at the two locations between May and October, 1965:

	1	2	3	4	5	6	7	8	9
Station A (Upstream)	May 12	June 1	June 21	July 12	July 31	Aug. 20	Sept. 10	Oct. 2	Oct. 22
Station B (Downstream)	May 17	June 6	June 29	July 18	Aug. 5	Aug. 26	Sept. 14	Oct. 9	Oct. 29

In the following text, a combination of letters and numbers will be used to indicate station and date of collection, e.g., A5 designates collection 5, taken July 31, at sampling station A (upstream).

Since difficulties were encountered in sampling the bottom fauna by conventional methods, a sampler was constructed which proved suitable for the environment studied. The cylindrical sampler has a diameter of 10.2 cm, is 25.4 cm long, and has a 2 meter handle attached at an angle most suitable for scooping through the substrate. The apparatus is shown at the right in Figure 3.

The organisms were collected using the sampler as well as a 15.2 × 15.2 cm Ekman dredge. Four to five samples were taken at each collection and washed in a sieve-bottom pail in which the size of the screen was 169 meshes per square cm. Specimens were placed into glass vials and preserved in 70% ethyl alcohol. In most cases the organisms were identified to genus by the author. Representative individuals were then sent out to specialists for confirmation of identifications.

Fishes were collected at twenty-day intervals using gill nets and a seine. Two 22.9 meter gill nets of 6.4 cm stretch and 2 meters depth were used. The nets were usually set during the morning and checked periodically until about twenty fishes were caught. However, during spring run-off, a seine 14 meters long and 5 meters deep with 2 cm stretch was used to collect the desired number of fishes. All fishes taken were killed immediately and their lengths measured in the field. Standard length (from the tip of the snout to the proximal edge of the hypural plate) was taken to the nearest millimeter and weight to the nearest tenth of a gram. Scale samples were collected from the left side of the fish in the area between the dorsal fin and the lateral line.

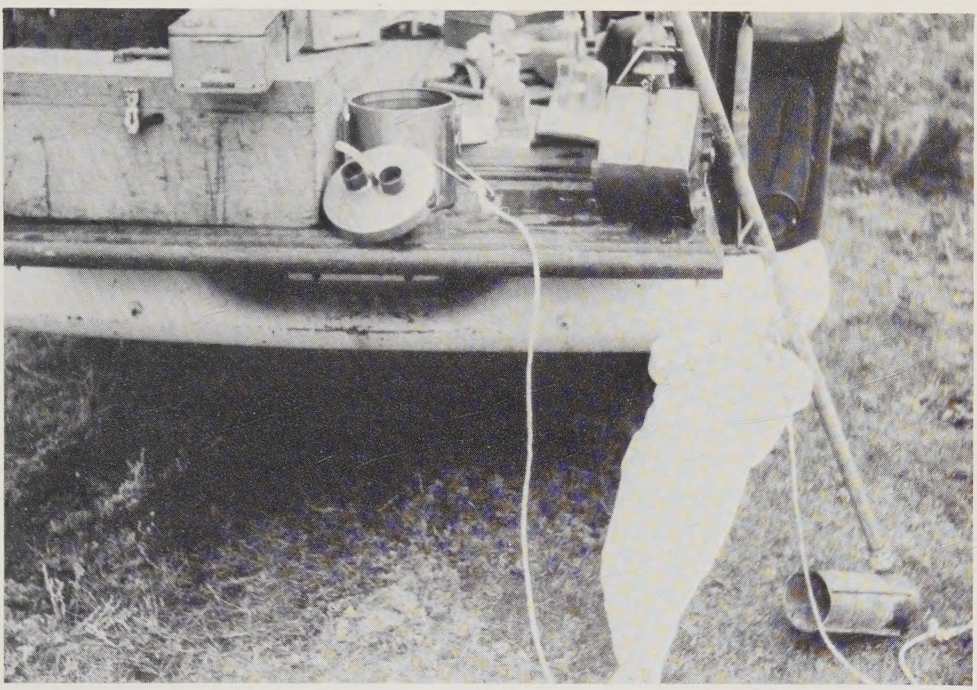

FIGURE 3. *Some of the equipment used in the field sampling with the handle of the bottom sampler standing next to the Ekman dredge.*

The age of the fishes was determined by examining the scales under a binocular compound microscope. Sucker scales are of the cycloid type with alternating zones of widely and closely spaced circuli. The annulus is marked by crossing over of the circuli and by their crowded condition. During late fall and winter, fish exhibit slower growth in response to unfavorable environmental conditions and an annulus is formed. In this study each annulus was accepted as indicating one year of growth.

Sex of the fishes was determined by examination of the gonads. This method proved successful except in inmature fishes whose sexual identity could not be determined because of weak gonadal development.

Contents of the stomach and/or intestine were placed into 10% formalin and taken to the laboratory. A combination of methods, as outlined by Hynes (1950), was tried to evaluate the stomach contents of the fishes. Because few suckers contained food in their stomachs these proved ineffective and only the relative abundance of certain food items in the entire digestive tract could be indicated. Data from this part of the study are, therefore, included for reference only.

With each fish and bottom fauna collection, a 24-hour water study was conducted involving hourly samples of dissolved oxygen (using the unmodified Winkler method), water and air temperatures (using a drybulb thermometer), as well as pH readings (using a portable Beckman pH meter). One quart samples of water were collected during the day at the highest pH reading and sent to the provincial analyst for the following analyses: total solids, ignition loss, hardness, sulphates, chlorides,

alkalinity, nature of alkalinity, nitrite and nitrate nitrogen, and phosphates. Hourly flow sheets in cubic feet per second (c.f.s.) were obtained from the Calgary Power development at Bearspaw Dam located immediately upstream from Station A and controlling much of the flow of water.

Fishes for the survey of parasites were collected on July 28, 29 and 30. A six meter seine of two cm stretch was used for these collections. Fishes from the Bow River as well as two tributary streams were included. The dates and localities, as well as the numbers and kinds of fishes collected, are as follows:

Bow River (Bearspaw) July 28, 1965
Catostomus commersoni (Lacepede) 25; *C. catostomus* (Forster) 25; *Pantosteus platyrhynchus* (Cope) 27.

Bow River (Stier's Ranch) July 30, 1965
Catostomus commersoni 26.

Bow River (Carseland Dam) July 29, 1965
Catostomus commersoni 28; *Pantosteus platyrhynchus* 25.

Big Hill Creek, July 28, 1965
Catostomus commersoni 25; *C. catostomus* 2.

Highwood River July 29, 1965
Catostomus commersoni 27; *Pantosteus platyrhynchus* 25.

The collections included three localities in the Bow River, each about 25 miles apart. The other two localities were Big Hill Creek, which empties into the Bow about 20 miles upstream from Calgary, and the Highwood River which joins the Bow about 25 miles downstream from the city.

The fishes were separated in the field according to species and locality, and frozen in solid carbon dioxide (dry ice). At the end of each collecting day the fishes were transferred to a deep freeze in the laboratory.

Prior to examination, the suckers were allowed to thaw and their standard lengths recorded. A ventral incision was made and the intestine, stomach, liver and coelomic cavity were examined under 20 × magnification. The gills were also removed and examined for infestation with parasites. The incidence and intensity of infection for each species of fish were then recorded and tabulated. The nematodes were fixed in 70% ethanol and identified as wet mounts in lactophenol. Acanthocephalans were fixed in a mixture of alcohol-formalin-acetic acid and identified in the unstained condition.

Results

Bottom Fauna

Appendix I shows the relative abundance of the organisms collected during the summer of 1965. Identifications were made to species in a number of instances. However, where only the generic name is included, these collections could represent one or more species.

It can be seen from Figure 4 that the number and diversity of invertebrate genera at Station A far exceeded those at Station B. Organisms belonging to 192 genera were collected in the upstream site compared to 83 genera found downstream. Chironomid larvae were the most abundant organisms at A. Table I shows the number of specimens and genera found in three sub-families of Chironomidae at the two stations.

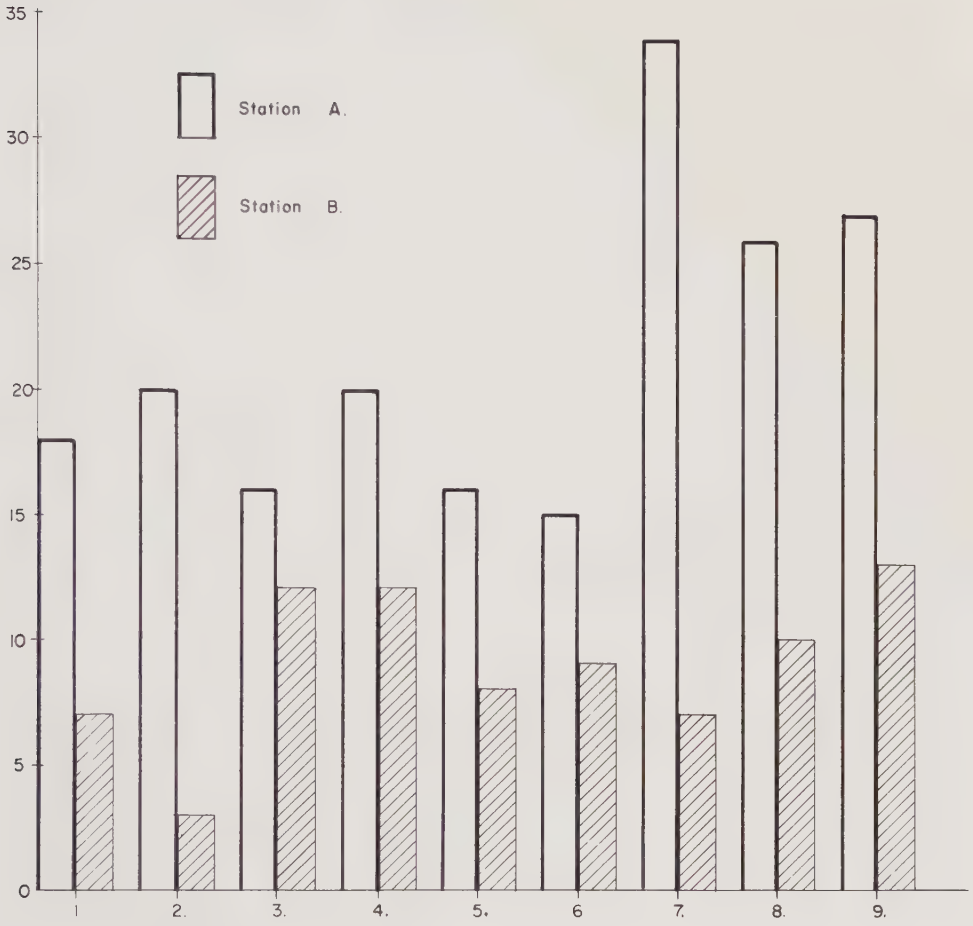

FIGURE 4. *Relative number of invertebrate genera from Stations A and B.*

At Station B the number of worms referable to the Tubificidae far exceeded that of any other organism. Table II shows the number of specimens and genera of tubificid species found at Stations A and B.

Table I

The number of genera and specimens found in the three subfamilies of Chironomidae at Stations A and B.

Subfamilies	No. of genera		No. of specimens	
	A	B	A	B
Chironominae	11	4	471	42
Orthocladiinae	6	4	58	27
Tanypodinae	3	2	114	34
Totals	20	10	643	103

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Table II

The relative numbers of tubificid worms found at each station.

Station	A	B
Tubifex tubifex	79	198
Limnodrilus hoffmeisteri	1	35
L. udekemianus	0	8
L. profundicola	0	3
L. claparedeanus	0	2
Limnodrilus sp. (immature)	3	18
Totals	83	264

Age-Condition

A total of 401 catostomid fishes was collected during the course of this study. A statistical analysis using the coefficient of condition K_{SL} ($K = W \times 10^5/SL^3$, where SL = standard length in centimeters and W = weight in grams) was made on all fishes. The condition factor is an expression derived from the length-weight relationship of fishes which approximates the cube law W/L^3, and is often used to compare the relative well-being of fishes under different environmental stresses. Table III shows the numbers of fishes in each age group, as well as the average coefficients of condition ($\bar{X}_K$), lengths ($\bar{X}_L$), and weights ($\bar{X}_W$) of the suckers collected at the respective sampling areas.

Table III

Numbers of fishes collected, age groups, K-values, lengths, and weights

Age (years)	*Catostomus commersoni*							
	Station A				Station B			
	No.	$\bar{X}_L$	$\bar{X}_W$	$\bar{X}_K$	No.	$\bar{X}_L$	$\bar{X}_W$	$\bar{X}_K$
2	10	9.1	15.0	1.97	1	11.5	42.5	2.77
3	4	14.5	64.6	2.07	0			
4	20	21.6	211.2	2.09	5	21.4	218.9	2.22
5	23	24.6	317.7	2.11	8	24.3	312.8	2.20
6	26	28.6	486.4	2.11	13	26.9	466.2	2.34
7	26	31.7	671.9	2.09	82	33.9	891.3	2.28
	109				109			

Table III (*Continued*)

Age	*Catostomus catostomus*							
	Station A				Station B			
	No.	$\bar{X}_L$	$\bar{X}_W$	$\bar{X}_K$	No.	$\bar{X}_L$	$\bar{X}_W$	$\bar{X}_K$
3					5	16.4	95.2	2.06
4	1	21.5	180.5	1.81	9	22.0	217.5	2.03
5	4	25.4	292.4	1.79	11	25.0	309.9	2.02
6	6	29.6	469.6	1.81	20	28.7	469.2	2.00
7	87	32.3	616.6	1.81	34	34.8	834.5	1.96
8	1	38.0	1013.8	1.85	5	41.3	1355.9	1.92
	99				84			

The K-value is often used to indicate suitability of an environment for fish species. Figure 5 shows the relationship between average K_{SL} values and age groups.

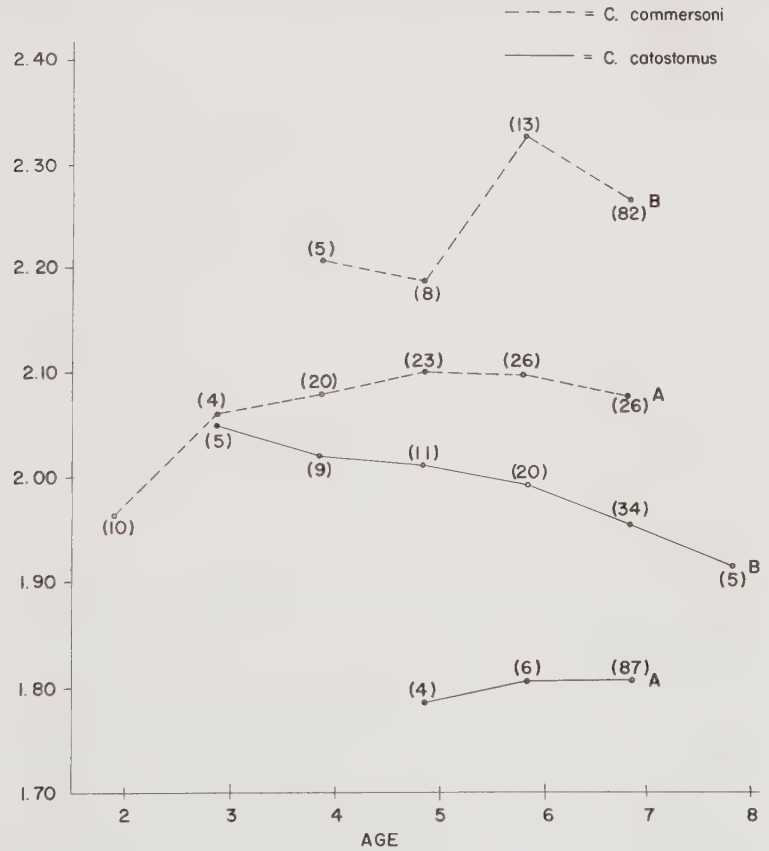

FIGURE 5. *Relationships of average K-values with age groups of catostomid fishes (number of fishes per age group in parentheses).*

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Since most of the sample sizes for the various age classes were very small and unequal (Figure 5), the seven-year old class is the only one in which an accurate comparison between the coefficient of condition of the fishes can be made. The student's "t" test at the 0.01 level of significance was applied to the average K_{SL} values obtained from the respective populations of seven-year old suckers. The results were as follows:

Table IV

Statistical analyses of K-values.

Fish Populations	N_B N_A	Level of Significance	Test	Critical "z" or "t"	Obtained "z" or "t"
Catostomus catostomus.....	34 87	.01	1-tailed	+2.33	+4.69
Catostomus commersoni.....	82 26	.01	1-tailed	+2.39	+4.87

Values from Table IV indicate that a significant difference exists at the 99% level in the condition of the groups of fishes as reflected by the samples. The fishes below Calgary appear in better condition than those above Calgary.

Food Studies

Difficulties were encountered in the food study because most of the food organisms were found in the intestine posterior to the pyloric sphincter. The softer tissues of these organisms were partially digested making identification difficult or impossible; however, parts of the exoskeletons of chironomid larvae, pupae and adults were noted in all fishes examined. Hemiptera, found in large numbers, were the only group in which a correlation could be established between seasonal occurrence in the bottom fauna collections and the food analysis.

Water Quality Studies

Water samples were collected on each sampling date. Tables V and VI show the results of the chemical analyses of the river water from Stations A and B. Concentrations are expressed in parts per million.

Total solids are often used as a measure of turbidity and can be determined by evaporating a quantity of filtered water at low temperatures. Sawyer (1960) indicates that domestic sewage, industrial wastes and street-washings contribute much organic and inorganic turbidity. The three factors mentioned by Sawyer probably account for the fact that turbidity was higher for nearly all collections at Station B. However, it can be noted from Figure 6 that an inverse relationship exists between turbidity and water flows.

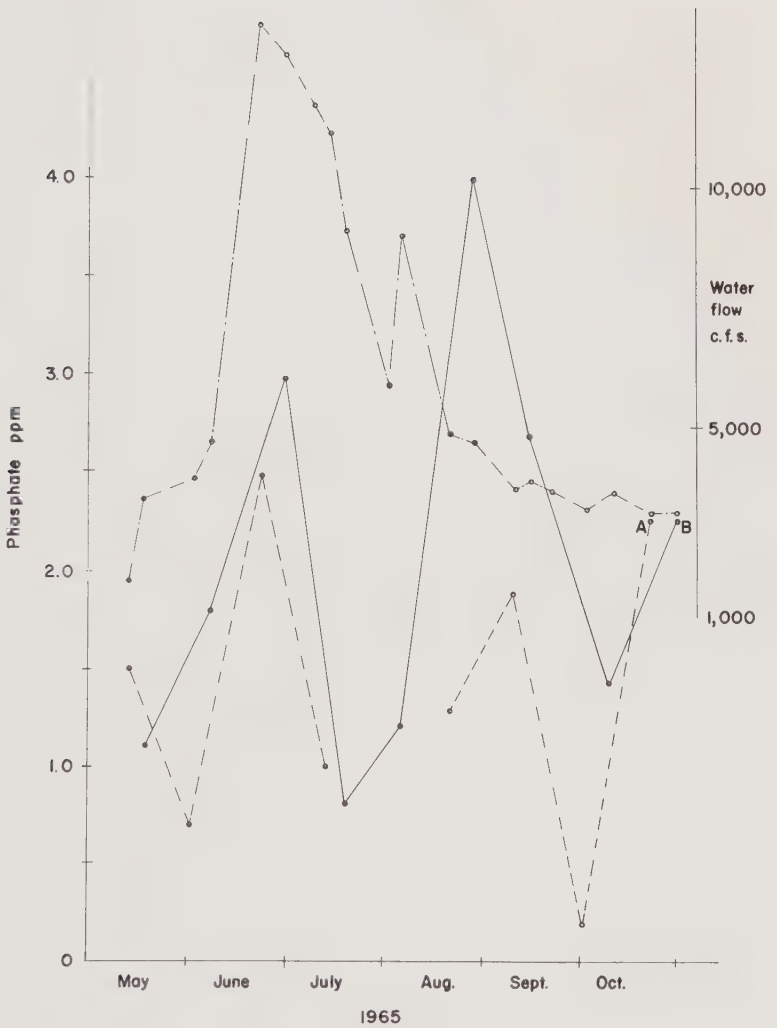

FIGURE 6. *Turbidity at Stations A and B, and water flow at Bearspaw Dam.*

Nitrogen analyses often are of value in stream pollution data because of the fertilizing effect of nitrogenous compounds on natural waters. Appreciable amounts of organic nitrogen are added to rivers and lakes by industrial and domestic effluents. The results from the water analyses indicate that nitrogen compounds are present in small quantities at Station B but virtually absent at Station A.

Phosphorus, another nutrient of great importance in fertilizing aquatic environments, is often considered the most critical single factor in the maintenance of biogeochemical cycles; Sawyer (1960) sets the critical level at 0.01 mg/1. The phosphate level at both stations was well above the previously-mentioned critical value. It can also be noted from Figure 7 that most of the water samples at Station B showed a higher phosphate level than those at Station A.

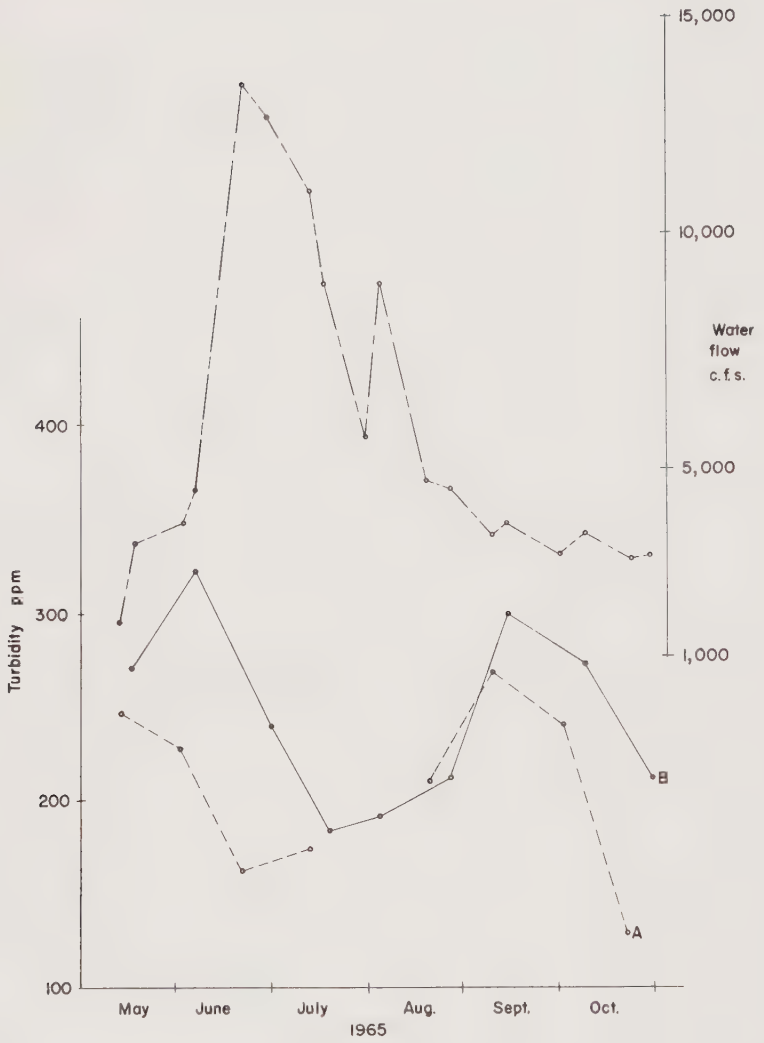

FIGURE 7. *Relationships between phosphate concentrations at Stations A and B and water flow at Bearspaw Dam.*

Table VII shows the ranges of the measurements of dissolved oxygen, temperature and pH (taken on an hourly basis, over a 24-hour period) obtained at Stations A and B.

Water temperatures at the two stations did not differ greatly and the pH remained relatively constant and slightly basic.

Concentration of oxygen in water is often expressed as the percentage saturation at normal pressure and at the temperature in question. Since Calgary has an average barometric pressure of approximately 669 mm of mercury (Rolston, 1962), the saturation level for various water temperatures was corrected to the average Calgary barometric pressure by a factor of 1.13. Water samples at Station A contained a relatively

Table V

Results of chemical analyses of water at Station A.

Collection No.	1	2	3	4	5*	6	7	8	9
Total solids	248	230	164	176	—	210	268	242	130
Ignition loss	44	94	60	70	—	84	92	84	74
Hardness	160	165	135	210	—	80	246	330	80
Sulphates	70	29	7	23	—	26	54	42	17
Chlorides	0	0	0	0	—	0	0	0	0
Alkalinity	125	165	150	115	—	100	130	150	145
Nature of Alkalinity	bicarbonates of lime and magnesium								
Nitrite nitrogen	0	0	0	0	—	0	0	0	0
Nitrate nitrogen	0	0	0	0	—	0	0	0	0
Phosphate	1.5	0.7	2.5	1.0	—	1.3	1.9	0.2	2.3

*The container of this sample was broken in transit, therefore no data were available for this date.

Table VI

Results of chemical analyses of water at Station B.

Collection No.	1	2	3	4	5	6	7	8	9
Total solids	272	324	240	184	192	212	302	274	212
Ignition loss	42	76	106	70	102	90	122	108	98
Hardness	170	170	150	160	200	140	345	245	210
Sulphates	83	97	39	30	16	20	56	44	16
Chlorides	3	8	0	0	0	0	0	0	0
Alkalinity	140	140	115	110	110	125	135	145	145
Nature of Alkalinity	bicarbonates of lime and magnesium								
Nitrite nitrogen	t	t	t	t	t	0	0	0	0
Nitrate nitrogen	0.2	t	1.5	0.3	1.8	0	1.7	0	0
Phosphate	1.1	1.8	3.0	0.8	1.2	4.0	2.7	1.4	2.3

t = trace

stable concentration of oxygen tending toward saturation. Station B showed increased diurnal-nocturnal oxygen fluctuation, and values at times were well below the saturation level. The lowest value obtained in 1965 was 6.0 ppm (6.7°C, 51% saturation).

Water flows are recorded by the staff of the hydroelectric plant at Bearspaw Dam, directly above Station A. From their data the averages of the flows during the summer months of 1965 were tabulated (Figure 8). Monthly flows in the summer of 1965 exceeded those for the summer of 1963 by an average of 1070 c.f.s.

Table VII

Ranges of temperature, pH, O_2, and % O_2 saturation at Stations A and B for the Summer of 1965.

Coll. No.	Air Temp °C	Water Temp °C	pH	O_2 (ppm)	%O_2 Sat.
Station A					
1	4.0–18.5	7.5–13.5	7.6–9.2	9.3–12.3	83–120
2	4.5–18.0	11.0–15.0	7.9–9.2	8.9–10.1	84–102
3	1.7–13.9	10.0–13.8	8.2–9.5	9.0–10.0	83–100
4	6.7–20.0	14.0–16.8	8.0–8.9	8.5– 9.2	86– 95
5	1.0–28.9	16.5–20.2	8.1–9.0	8.2– 9.2	87–105
6	6.1–27.8	15.1–17.2	7.8–9.0	8.2– 9.6	85– 93
7	0.0–12.2	9.2–11.2	7.6–8.6	9.3– 9.9	85– 93
8	−1.4–16.6	6.3– 9.1	7.9–8.6	10.0–10.4	85– 93
9	−0.9–16.5	4.4– 8.3	8.0–8.2	10.4–10.9	83– 97
Station B					
1	−5.0–16.8	7.0– 9.5	7.0–6.8	7.9–11.0	68– 98
2	1.7–20.0	10.0–14.2	8.0–9.4	6.8–10.5	65–106
3	5.6–11.1	10.5–11.8	7.1–8.7	8.6– 9.6	83– 89
4	10.6–27.8	13.5–17.2	8.0–8.7	7.8– 8.3	79– 87
5	9.5–26.1	15.7–18.2	7.9–9.1	7.6– 8.3	79– 90
6	7.2–25.0	14.1–18.1	7.9–8.9	6.2– 8.0	65– 91
7	−1.7– 0.5	6.3– 9.1	7.7–8.8	7.2–12.4	63–111
8	−2.5–11.1	6.1– 8.6	7.7–8.4	7.2–10.6	60– 91
9	−0.9–11.3	5.5– 8.0	7.6–8.4	6.0–13.7	51–118

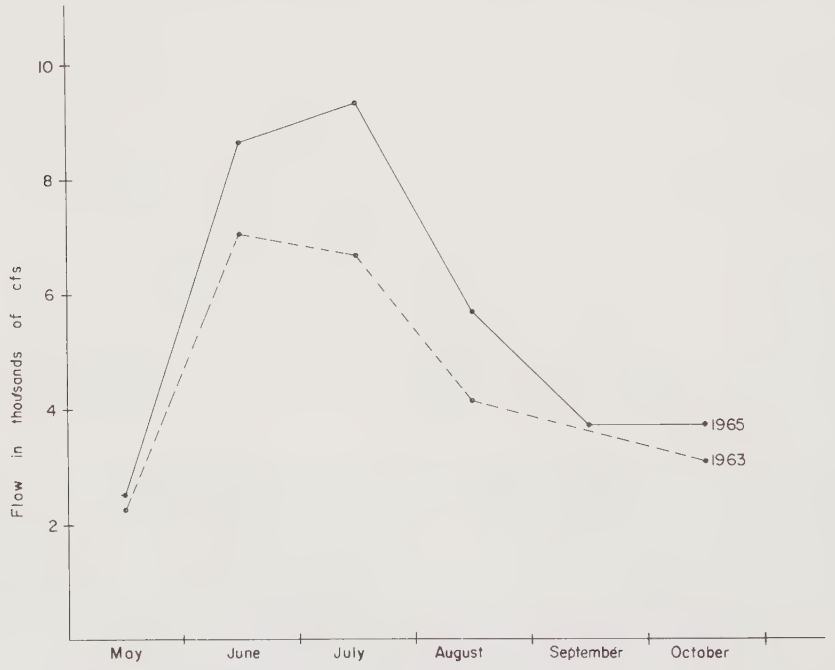

FIGURE 8. *Average summer flows past Bearspaw Dam, 1963 and 1965.*

Parasitological Studies

The parasitological examinations indicated a difference in distribution and abundance of parasites in the three species of suckers. Parasites found included two species of Acanthocephala, *Octospinifer macilentus* Van Cleave, 1919, and *Neoechinorhynchus cristatus* Lynch, 1936, as well as a nematode *Rhabdochona fortunatowi* Dinnik, 1933; the latter is referable to the family Spiruridae. Table VIII shows relationships of locality, intensity and incidence of infection and average length (cm) of host specimens.

Table VIII

Relationships of parasite infection with length of host and place of capture.

	Fishes		Parasites					
			Incidence			$\bar{X}$ Intensity		
	No.	$\bar{X}$ Length	1	2	3	1	2	3
BHC								
C. commersoni	25	12.07	16	16	0	7.4	6.6	0
C. catostomus	2	13.00	0	0	0	0	0	0
BBD								
C. commersoni	25	10.27	13	17	0	6.5	8.5	0
C. catostomus	25	9.34	0	3	3	0	1.0	2.0
P. platyrhynchus	27	8.40	0	0	1	0	0	2.0
BSR								
C. commersoni	26	6.11	0	3	0	0	1.0	0
HR								
C. commersoni	27	4.32	0	0	0	0	0	0
P. platyrhynchus	25	6.60	0	0	0	0	0	0
BRC								
C. commersoni	28	5.42	0	1	0	0	1.0	0
P. platyrhynchus	25	6.56	0	0	0	0	0	0

Legend: 1—*Octospinifer macilentus;* 2—*Neoechinorhynchus cristatus;* 3—*Rhabdochona fortunatowi*
BHC—Big Hill Creek; BBD—Bow River, Bearspaw Dam,
BSR—Bow River, Stier's Ranch; HR—Highwood River;
BCD—Bow River, Carseland Dam.

The incidence of infection is shown by simply indicating the percentage of fishes infected. Average intensity of infection is calculated by dividing the total number of parasites by the number of fishes infected. A heavy infection with Acanthocephala can be noted in *C. commersoni* taken at Big Hill Springs and Bearspaw, but not at the other localities. Attention is also drawn to the fact that intensity of infection seems to be correlated with fish size. The *C. commersoni* collected at Big Hill Creek and

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Bearspaw Dam had an average length of 12.07 cm and 10.27 cm respectively and showed a relatively high intensity of Acanthocephala infection. *Catostomus commersoni* from the Highwood River with an average length of 4.32 cm were not infected with Acanthocephala.

C. catostomus and *P. platyrhynchus* showed relatively light parasitic infections at the five collection sites. The nematode infection was of very low intensity with an average of approximately one worm per infected fish.

Discussion

The great variety of the number and kinds of invertebrates collected at the two stations is indicated by the tables in Appendix I. At Station A, large numbers of genera, each represented by relatively few individuals, were present. Station B was characterized by few genera, but a relatively high abundance of individuals. Some of the groups present at Station A but absent from Station B were Plecoptera, Ephemeroptera, Trichoptera, and some dipteran families. These findings are in accordance with Hynes (1960) who indicates that when organic pollution is fairly severe, the effects on most invertebrates are so marked that the environment of whole taxonomic groups, rather than individual species, is affected. Further evidence indicating pollution are personal observations made in 1963 and 1964; at those times Ephemeroptera, Plecoptera, and Trichoptera nymphs were present in portions of the river approximately 30 miles downstream from Station B. Since the effects of pollution decrease with increasing distance downstream, assuming, of course, that no additional pollution is occurring, the river had presumably "purified" itself enough to enable the nymphs to remain established at these locations. The assumption is made that dilution of effluents, sedimentation of particulate matter and biological oxidation of organic materials all play a significant role in returning a river to a condition approximating its "normal" state.

The tubificid worm, *Tubifex tubifex*, made up the bulk of each collection at Station B. Although the oligochaete was also present in the collections at Station A, there was only half the number at the latter site. Brinkhurst (1962, 1965) tentatively suggests that *Tubifex tubifex*, *Limnodrilus hoffmeisteri*, as well as *L. udekemianus*, are quite resistant to organic and inert mineral pollution. However, in 1965 Brinkhurst also pointed out that species often considered to be pollution-indicators, may be found in quite normal situations and that *L. hoffmeisteri* is, in fact, one of the most cosmopolitan and ecologically tolerant animals in fresh waters. This observation is substantiated in the present study with the presence of T. *tubifex* in collections from each of the two stations. However, the important observation is that *T. tubifex* was more than twice as abundant at Station B as at A. Data of this nature have also been obtained by Hynes (1960) and Patrick (1949), who indicate that the few species surviving pollution do so as a result of less competition for habitat and food and fewer predators.

Often cited to be associated with and occurring in conjunction with pollution are some members of the family Chironomidae. The dipteran larvae most frequently reported from polluted waters are those referable to the genus *Chironomus* (Hynes, 1960; Carr and Hiltunen, 1961). These findings are in accord with the results of the

collections at Station B where 10 different chironomid genera were collected and at Station A with a total of 20 genera. *Chironomus* was the most abundant genus at B, and *Stictochironomus* at A. The point to be emphasized is not the absence or presence of one particular species, but the relatively disproportionate abundance of the chironomid genera at the respective stations.

Although the floristic components of aquatic habitats are frequently neglected in pollution studies, these organisms can be quite useful in assessing water quality. Williams (1964) reported that an intensive study of plankton from more than 100 stations did not reveal any one species or any small groups of species whose mere presence may be regarded as an indication of degree of water quality. He suggested that relative numbers or biomasses of any kind rather than particular kinds must be used for water quality studies. Plankton studies carried out by the author in areas upstream and downstream from Calgary in 1964 showed that numbers and kinds of planktors differed at these sites. However, because high-water flows often make plankton studies meaningless (Williams, 1964), this aspect of the present study was deleted. Observations of the macroflora indicated that *Potamogeton* (probably *P. pectinatus*) was abundant during the late summer at Station B. This macrophyte was observed to cover completely the river bottom as far downstream as thirty miles below the Bonnybrook Sewage Outfall. The same observations were recorded during the spring in previous years (Rolston, 1962; Kussat, 1966). The abundant plant growth reflects increased fertility of the habitat presumably effected to a large extent by the addition of nitrite, nitrate and phosphate to the river. As can be noted in Tables V and VI, nitrates and phosphates are present in larger quantities at Station B than at Station A. Sawyer (1960), Hynes (1960) and Ferguson (1965) indicate that the above-mentioned nutrients have been reported in large quantities from direct measurements of industrial and domestic effluents; Williams (1964) also adds that the products of secondary treatment of domestic sewage are enriching substances because they contain mineralized products such as nitrates and phosphates.

Ferguson (1965) lists the following factors as major contributors to aquatic weed growth in streams: abundance of nutrients, temperature, intensity of light and velocity of water. Personal observations indicate that in recent years the above-mentioned factors could have been instrumental in aiding the growth of *Potamogeton*. These plants were well established in the river below Calgary as early as May 15, 1963, and May 23, 1964, and appear to have a decided influence on the dissolved oxygen levels of the river. Observations on the minima and maxima during previous summers downstream from Calgary revealed that the dissolved oxygen content fluctuated from a seasonal high of 13.6 ppm in the afternoon to a low of 2.7 ppm during the early morning (Rolston, 1962). In May, 1963, oxygen content analyses of the Bow River showed a maximum value of 15.3 ppm (16°C, 160% saturation) and a minimum of 4.1 ppm (12.9°C, 40% saturation) at Stier's Ranch (Kussat, 1966).

In contrast to previous data (Rolston, 1962; Ferguson, 1965; Kussat, 1966), the pattern of oxygen concentrations at Station B in 1965 did not show a great degree of diurnal-nocturnal fluctuations (Figures 9 and 10). This observation, when considered along with the fact that the pond weeds did not appear until late in the summer, give some indication that the oxygen fluctuations are related, at least in part, to the photosynthetic activity of *Potamogeton*, and thereby substantiate Rolston's (1962)

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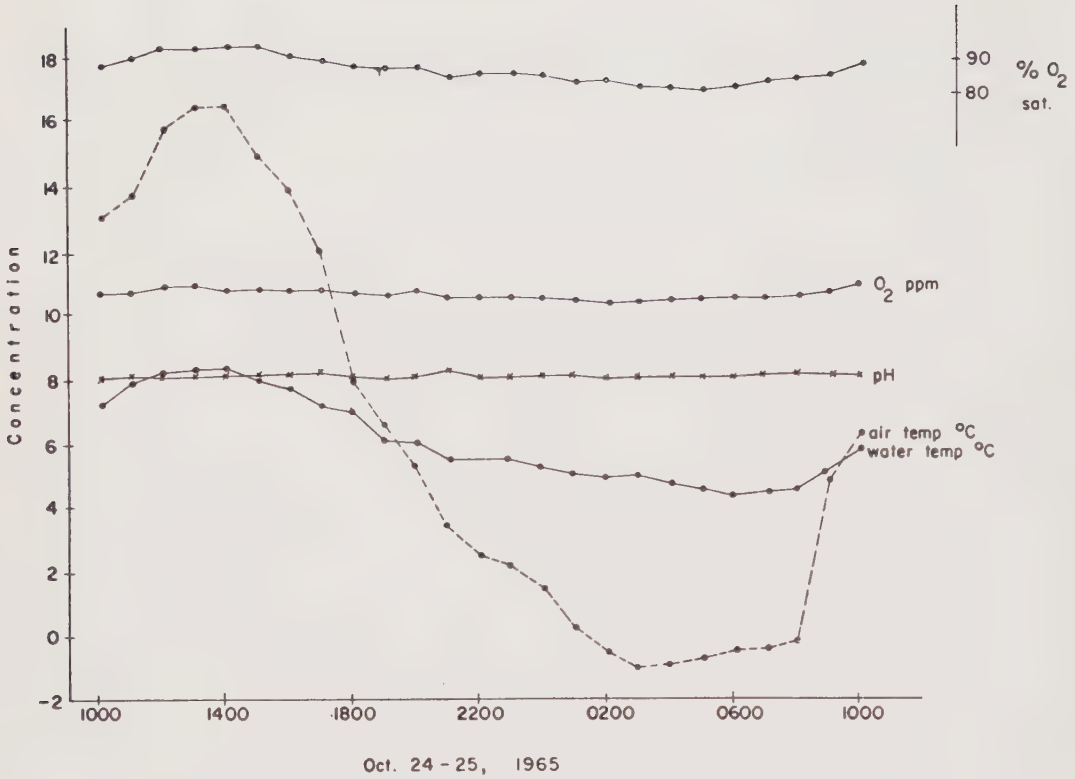

FIGURE 9. *Oxygen, pH and temperature relationships at Station A.*

statement that low nocturnal-dissolved oxygen levels are caused by aquatic plant growth. The absence of the aquatic macrophytes in the early summer of 1965 can probably be related to the increased water-flows and consequent turbidity and lower nutrient content of the water at that time (Figure 8). Williams (1964) points out that total dissolved solids are often lower in streams during high flows caused by surface runoff. This condition can be directly related to the dilution of organic solutes and the fact that the stream contains proportionately less groundwater which usually has a higher dissolved mineral content than surface runoff. The increased flows during 1965 were the result of an unusually heavy runoff from the mountains into tributary streams which created turbid conditions for longer periods than in previous years. Effects upon the dissolved oxygen regime by plants that did grow under these conditions were probably held in check by the constant sedimentation of silt and the consequent low intensity of light available for photosynthesis.

Because of the importance of the Bow River as a fishing stream, the Provincial Departments of Lands and Forests and Public Health have deemed it necessary that the dissolved oxygen content in the river be kept at a level of 6 ppm or higher in order to support fish life. This value generally compares favorably with those cited in the literature. Patrick (1949) indicates standards of dissolved oxygen for a polluted stream as follows: monthly average over 4 ppm, daily minimum 3 ppm. Lagler (1959)

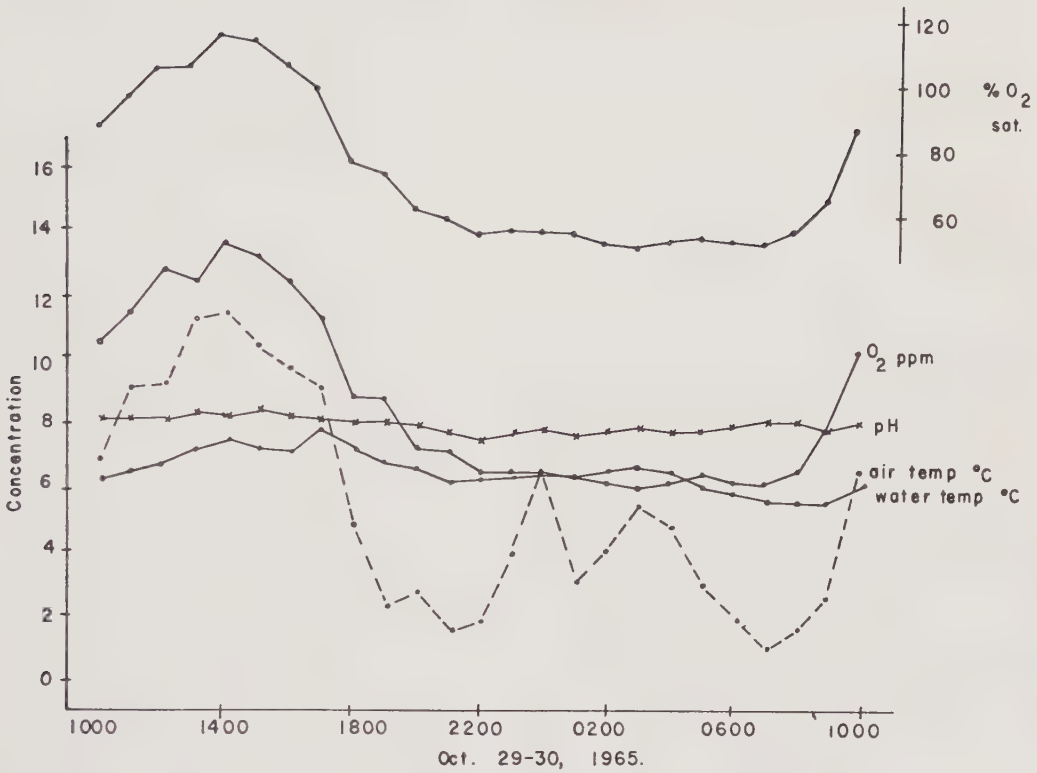

FIGURE 10. *Oxygen, pH and temperature relationships at Station B.*

cites that the Aquatic Advisory Committee (1955) has redefined the dissolved oxygen criterion as follows: "The dissolved oxygen content of warm water fish habitats shall be not less than 5 ppm during at least 16 hr. of any 24 hr. period; but at no time shall the oxygen content be less than 3 ppm". Coldwater fishes are apparently more susceptible to low oxygen tensions than warm water fishes; Davis (1961) indicates that trout should not be held for any length of time in water with an oxygen content of less than 5 ppm. Although limiting oxygen levels for different species of fishes are frequently cited in the literature, the author feels that these values, often established under artificial conditions and with hatchery reared fish, cannot be applied to established populations living in a natural environment. As mentioned previously, dissolved oxygen levels in the Bow River reached a low of 2.7 ppm in 1962, but no obvious detrimental effects on the fauna were observed. However, since the periods of low oxygen content are relatively short and have presumedly occurred over a number of years, the fishes probably compensate for the environmental stress through physiological and behavioral adjustments. Responses of this nature have been documented by Rounsefell and Everhart (1953) who reported that fish were found living in a healthy condition at extremely low oxygen concentrations and that these fish showed a physiological compensation through a high red blood cell count.

The relative abundance of fishes in their respective age groups indicates that selective sampling did occur. As a result of this selectivity, no evaluation of the catostomid population structure could be made, and the comparisons of K-values were restricted to one age group only. The poor representation of the younger groups can be traced directly to the size of mesh used for fishing. Because the 2 cm mesh size used in this study caught large numbers of seven-year old fish, it is safe to assume that the larger five and six-year old suckers, as well as the small eight-year olds, were also selectively netted by this mesh size. The average measurements for the five and eight-year old fishes would then be strongly biased and unsuitable for statistical treatments. For these reasons only the seven-year old class was used for the analysis of the populations at Stations A and B. Selective netting was also pointed out by Spoor (1938) who states that, depending upon the size of the mesh and consequent selective action of these mesh sizes, often only the larger and more rapid growing suckers of the next lower year class are captured.

In the present study, comparisons of the seven year old suckers indicated that the calculated coefficient of condition of fishes at Station B was higher than that at A. Related studies in the same geographical areas support this observation in that corresponding populations of a hybrid trout (*Salmo gairdneri* × *clarki*) also showed significantly higher K-values when taken downstream from Calgary (Kussat, 1966). Although beneficial effects from sewage at the present time appear responsible for the better "condition" of fish below Calgary, this is probably only a temporary phenomenon. An increase in the pollution load of the river could expose the aquatic organisms to ever-increasing environmental stresses, e.g., fish exposed to lower oxygen concentrations over longer periods of time, would eventually reach their compensatory limit and be eliminated.

Because the bottom fauna studies revealed large differences in the numbers and kinds of organisms found at each station, it seems sound to assume that utilization of these animals by the fishes also differed at the two localities. It can be seen from Table II that tubificid worms were present in large numbers at Station B but rare at Station A. These sedentary worms conceivably provide a ready food source for the bottom feeding suckers in downstream areas. On the other hand, much of the bottom fauna at Station A was represented by organisms known to be highly active, i.e., Plecoptera, Ephermeroptera, Coloeptera, and thereby less susceptible to sucker predation.

Stewart (1926) indicates that although examinations of the stomach contents of the white sucker have been made by various workers, much disagreement exists about the feeding habits of these fish. The present results obtained from the stomach analyses of the suckers proved inadequate to speculate on the types of food utilized by the two species of fishes; however, it was noted that *C. commersoni* was often caught at different times during the fishing period than *C. catostomus*. The separation in time of the activity of the two species would then allow them to coexist in a given habitat.

Competition with and predation by other fish groups inhabiting the same waters probably also contribute much to the disparities of the K-values of the catostomid fishes. At Station A, five groups of carnivorous fishes were netted in conjunction with the suckers. Limited examination of the stomach contents of *Lota lota*, *Esox lucius*,

Salmo gairdneri × *clarki*, *Salvelinus namaycush* and *Salmo trutta* indicated the presence of young suckers in their diet. This predation on the young suckers at Station A conceivably has a beneficial effect by decreasing population size and enabling *Catostomus commersoni* and *C. catostomus* to coexist in a fairly restricted environment. Similar observations were recorded by Fryer (1959) who states that predators, by controlling the population density, will limit competition and therefore favour the survival of certain species. At Station B, the increased yield of a limited number of species of bottom organisms apparently has removed competition allowing the suckers to exist in a similar habitat.

The Rocky Mountain whitefish, *Prosopium williamsoni*, was absent from the lower test sites but was present in large numbers at Station A. This fish, known to frequent oligotrophic waters, probably competes with the suckers for certain food items because of its similar external morphological characteristics, e.g., mouth parts. At Station B, the only fishes collected, other than catostomids, were *Salmo gairdneri* × *clarki*. Therefore, it is assumed that interactions with only a single species of fish placed less stress on the sucker populations at Station B thus allowing the latter fishes to utilize more energy for maintenance and growth. The marked absence of oligotrophic fishes at B strongly suggests that environmental conditions might have been degraded sufficiently to exclude all but the hybrid trout from these polluted waters.

In contrast to the food analyses, significant results from the parasitological survey indicated that the intensity and incidence of infection of the catostomid fishes at Stations A and B varied. *Catostomus commersoni* taken above Calgary was shown to be heavily infected with the acanthocephalans, *Octospinifer macilentus* and *Neoechinorhynchus cristatus*, but these fish showed a relatively light infection at collection sites downstream from the city (Table VIII). These results can be readily explained by examining the life cycles of the parasites as reported in the literature. The intermediate hosts which can be utilized by the above-mentioned acanthocephalan species are amphipods or ostracods. Since analyses of the bottom fauna indicated that these food organisms were present in the upstream but absent in downstream study areas, it is apparent that the polluted water below Calgary represents a barrier to these parasites by the exclusion of their intermediate hosts. Accepting this explanation, one would then expect the parasites to re-occur at some point downstream where the effects of pollutants have been dissipated sufficiently to allow the intermediate hosts to exist. This reasoning has been supported by Sloan (per. comm. 1966) who has examined suckers taken at Bassano Dam. He found that fishes at the latter locality were commonly infected with the same acanthocephalan species as those recovered at Bearspaw Dam.

Host-parasite relationships have in the past been used in solving many ecological problems. Dogiel (1962) indicates that the mode of feeding and the type of diet determine to a great extent the composition of the parasite fauna of the host. He states further that parasites as indicators of diet are among those clues which allow us to make deductions from the type of the parasite fauna about various aspects of the ecology of the host. In the present study, all three host species examined were collected within a limited period of time to minimize the differences which could result from seasonal variation in the parasitic fauna. In consideration of this precaution and since *C. commersoni* was the only form heavily parasitized, it is suggested that the

three species of fishes have different feeding habits, i.e., to the extent that those components of the fauna used as intermediate hosts are not utilized in common. This observation seems to coincide with a competition study carried out by Maitland (1965). He points out that although the five sympatric species of fishes (*Salmo salar*, *Salmo trutta*, *Phoxinus phoxinus*, *Nemacheilus barbatula*, *Gasterosteus aculeatus*) which were studied have important food items in common, none has an absolute identity of diet.

Conclusions

Definite environmental differences were noted between Stations A and B along the Bow River, Alberta. Comparative studies including bottom fauna analyses, age and size relationships of fish, physical and chemical water analyses, all attest to the disparity in the environment afforded by the two backwashes. Industrial and domestic effluents piped into the Bow River as it passes through Calgary are probably the primary cause for the dissimilarities at the two stations.

Results of the parasitological survey strongly suggest the distributional pattern of the intestinal parasites *Octospinifer macilentus* and *Neoechinorhynchus cristatus* found in *Catostomus commersoni* can be directly related to the exclusion of their intermediate hosts from the "polluted" waters. For the fishes considered in this work, the increase in biomass in the downstream areas has been shown to be at least temporarily beneficial to two species of suckers and hybrid trout to the extent that growth, in terms of weight and length, appears to be favoured by the addition of effluents. Although hybrid trout appear abundant at this time, further eutrophication could result in the elimination of all salmonid fishes from large portions of the Bow River. Also, the use of the natural resources has been minimized in terms of recreation in that present conditions have restricted water sports, reduced water quality and degraded the culinary quality of fish.

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Appendix

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total A	Total B
Diptera																				
Chironomidae																				
Chironominae																				
Harnischia	—	—	3	—	17	1	3	—	10	—	15	—	5	—	—	—	—	—	53	1
Phaenopsectra	—	—	1	—	—	—	—	3	10	—	—	—	25	1	12	—	20	—	68	4
Tanytarsus	—	—	4	—	18	—	20	—	39	1	4	1	10	—	2	—	43	—	140	2
Chironomus	4	—	—	—	—	—	—	—	—	—	4	—	4	12	1	14	—	9	13	35
Stictochironomus	30	—	5	—	—	—	—	—	—	—	—	—	20	—	36	—	50	—	141	0
Limnochironomus	—	—	—	—	—	—	1	—	1	—	—	—	5	—	4	—	6	—	17	0
Cryptochironomus	—	—	2	—	6	—	1	—	1	—	3	—	7	—	6	—	—	—	26	0
Paratendipes	—	—	3	—	—	—	4	—	—	—	—	—	1	—	—	—	—	—	8	0
Sergentia	—	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	0
Polypedilum	1	—	—	—	—	—	2	—	—	—	—	—	—	—	—	—	—	—	3	0
Microtendipes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	1	0
Diptera																				
Orthocladiinae																				
Cricotopus	3	7	3	3	—	—	3	—	4	—	—	—	1	—	1	—	—	2	15	12
Orthocladius	2	5	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	5
Diamesa	—	6	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	6
Rheocricotopus	—	—	10	—	2	—	2	—	10	—	2	2	3	2	5	—	—	—	34	4
Prodiamesa	—	—	2	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	3	0
Heterotrissocladius	—	—	—	—	—	—	—	—	1	—	1	—	—	—	—	—	—	—	2	0
Tanypodinae																				
Procladius	8	—	3	—	1	—	—	—	—	—	7	—	2	3	1	1	3	—	25	4
Psectrotanypus	—	—	1	—	—	—	4	4	8	—	—	—	48	—	20	21	6	5	87	30
Thienemannimyia group	—	—	—	—	2	—	—	—	—	—	—	—	—	—	—	—	—	—	2	0

Appendix *Continued*

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total A	Total B
Heleidae																				
Palpomyia	1	—	1	—	11	—	1	—	—	—	—	—	—	—	—	—	—	—	14	0
Tabanidae																				
Tabanus	—	—	—	—	1	—	1	—	—	—	1	—	—	—	3	—	—	—	6	0
Tipulidae																				
Tipula	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	1	0
Ephemeroptera																				
Baetidae																				
Serrata group	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	0
Simplex group	—	—	—	—	—	—	1	—	—	—	—	—	—	—	—	—	—	—	1	0
Caenis	4	—	6	—	2	—	5	—	—	—	—	—	2	—	4	—	11	—	34	0
Tricorythodes	—	—	—	—	—	—	—	—	—	—	1	—	1	—	—	—	—	—	2	0
Paraleptophlebia	—	—	—	—	—	—	—	—	—	—	—	—	1	—	2	—	—	—	3	0
Ephemeridae																				
Ephemera	—	—	—	—	—	—	—	—	1	—	—	—	—	—	1	—	4	—	6	0
Plecoptera																				
Chloroperlidae																				
Alloperla	2	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	0
Trichoptera																				
Limnephilidae																				
Limnephilus	2	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	3	0

Appendix *Continued*

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total A	Total B
Lepidostomatidae																				
Lepidostoma	—	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	0
Phryganeidae																				
Phrycanea	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	—	—	1	0
Coleoptera																				
Dytiscidae																				
Bidessus	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	1	—	2	0
Haliplidae																				
Brychius	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	1	—	2	0
Hemiptera																				
Corixidae																				
Callicorixa bifida	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	0	2
Callicorixa audeni	—	—	—	—	—	—	—	—	—	—	—	—	5	—	6	9	5	10	16	19
Sigara solensis	4	—	—	—	—	—	—	—	—	—	—	—	2	—	2	—	1	—	9	0
S. grosslineata	—	—	—	—	—	—	—	—	—	—	—	—	4	—	—	—	12	—	16	0
S. modesta	—	—	—	—	—	—	—	—	—	—	—	—	3	—	—	—	—	—	3	0
S. alternata	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	2	—	2	0
Cenocorixa bifida	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	4	—	4	0
C. dakotensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	0	1
Hydracarina																				
Limnesiidae																				
Limnesia	—	—	—	—	—	—	—	—	—	—	3	—	4	—	4	—	36	—	47	0

Appendix *Continued*

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total	
																			A	B
Hygrobatidae																				
Hygrobates	—	—	—	—	—	—	—	—	—	—	—	—	1	—	1	—	7	2	9	2
Hydracarina																				
Lebertiidae																				
Lebertia	—	—	1	—	—	—	—	—	—	—	1	—	—	—	—	—	6	—	8	0
Unionicolidae																				
Neumania	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	1	0
Pionidae																				
Piona	—	—	—	—	—	—	—	—	—	—	—	—	2	—	1	—	1	—	4	0
Hirudinea																				
Erpobdellidae																				
Nephelopsis obscura	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	0	2
Erpobdella punctata	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	2	0	3
Dina dubia	—	—	—	—	—	—	—	1	—	—	—	—	—	1	—	—	—	1	0	3
Glossiphoniidae																				
Glossiphonia complanata	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	—	1	1	1
Hellobdella stagnalis	5	—	—	—	—	—	1	—	—	—	—	—	1	—	1	—	—	—	8	0
Oculobdella lucidia	—	—	—	—	—	—	1	—	1	—	—	—	1	—	—	—	—	—	3	0

Appendix *Continued*

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total A	Total B
Oligochaeta																				
Tubificidae																				
Tubifex tubifex	7	30	3	19	12	20	20	27	1	7	9	20	12	15	3	30	12	30	79	198
Limnodrilus (immature)	1	—	—	—	—	2	—	—	—	—	—	—	1	18	1	—	—	—	3	20
L. hoffmeisteri	—	—	—	—	1	1	—	4	—	5	—	—	—	—	—	10	—	15	1	35
L. udekemianus	—	—	—	—	—	1	—	3	—	3	—	1	—	—	—	—	—	—	0	8
L. profundicola	—	—	—	—	—	1	—	—	—	2	—	—	—	—	—	—	—	—	0	3
L. claparedeanus	—	—	—	—	—	1	—	—	—	1	—	—	—	—	—	—	—	—	0	2
Naididae																				
Uncinais uncinate	—	—	—	—	—	—	3	—	1	—	—	—	—	—	—	—	—	—	4	0
Amphipoda																				
Talitridae																				
Hyalella azteca	—	—	—	—	—	—	—	—	—	—	—	—	2	—	—	—	1	—	3	0
Gastropoda																				
Physidae																				
Physa sayii	1	—	—	—	1	—	1	—	1	—	—	3	1	—	—	—	—	3	5	6
Succinaidae																				
Succinea	—	—	—	—	—	1	—	1	—	—	—	2	—	—	—	—	—	—	0	4
Planorbidae																				
Gyraulus parvus	7	—	—	—	—	—	—	—	—	—	5	—	—	—	—	—	—	—	12	0
G. circumstriatus	—	4	—	—	6	—	4	—	—	—	—	—	6	—	—	—	—	—	16	1
G. deflectus	—	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	0

Appendix *Concluded*

The relative abundance of invertebrate organisms collected at Stations A and B

Collection No.	1A	1B	2A	2B	3A	3B	4A	4B	5A	5B	6A	6B	7A	7B	8A	8B	9A	9B	Total A	Total B
Lymnaeidae																				
Lymnaea humilis	—	—	2	—	5	1	—	1	6	—	6	—	1	—	3	—	4	—	27	2
L. palustris	7	—	—	—	3	3	—	—	—	—	—	—	—	—	—	—	—	—	10	3
L. parva	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	9	—	9	0
L. catascopium	—	—	—	—	—	1	—	1	—	2	—	2	—	—	—	2	—	—	0	8
Pupillidae																				
Vertigo binneyana	—	1	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0	1
Valloniidae																				
Vallonia gracillicosta	—	—	—	1	—	1	—	1	—	—	—	1	—	—	—	—	—	—	0	4
Zonitidae																				
Zonitoides arboreus	—	—	—	—	—	—	—	1	—	—	—	—	—	—	—	—	—	—	0	1
Hawaiia minuscula	—	—	—	—	—	—	—	1	—	1	—	1	—	—	—	—	—	—	0	3
Pelecypoda																				
Sphaeriidae																				
Pisidium nitidum	—	—	—	—	3	—	—	—	—	—	—	—	1	—	—	—	4	—	8	0
P. compressum	—	—	—	—	—	—	2	—	2	—	3	—	—	—	—	—	3	—	10	0
Sphaerium lacustra	—	—	—	—	—	—	—	—	—	—	—	—	—	—	1	—	—	—	1	0

Loss of Petersen Disk Tags from Spawning Chum Salmon (*Oncorhynchus keta*)

by D. Brent Lister and Richard A. L. Harvey

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Abstract

Petersen disk tag loss from spawning chum salmon was studied over a three-year period in an artificial spawning channel. The time from tagging to recovery averaged approximately 11 days. Accurate enumeration of fish entering the channel and virtually complete recovery of dead on completion of spawning enabled estimation of tag loss on the basis of changes in the tagged:untagged ratio over the spawning period.

The average rate of tag loss from males (30%) was significantly higher than the loss from females (9%). Loss among males was also related to date of entry into the channel, with tags applied early in the run having a greater chance of loss than those applied late in the run. The addition of accessory buffer disks to the tag reduced loss by 50 percent, thus indicating that disk failure was the primary factor responsible.

Introduction

The mark and recapture method, involving the application of Petersen disk tags, is used extensively to estimate spawning populations of Pacific salmon. Mature salmon are tagged at some point on their spawning migration and representative tagged:untagged ratios are generally obtained from the examination of carcasses available on the spawning grounds. Population estimates based on tagging, when compared with fence counts, have been shown to have significant positive bias (Vernon et al., 1964). The magnitude of error has also been shown to be positively related to the distance and time period between tagging and recovery (Schaeffer, 1951). Ricker (1958) has presented a thorough review of the various causes of this type of bias. The purpose of this paper is to report on one of the sources of error, namely, tag loss due to shedding or failure of the tag.

Rates of Petersen disk tag loss occurring in salmon net fisheries have been measured in double mark experiments conducted by Bevan (1962) and Parker (1962). The failure of Petersen disk tags on the spawning grounds has been observed (Calhoun et al., 1951) but there is a lack of information on specific rates of loss from salmon in fresh water.

This study was undertaken over a three-year period at an artificial spawning channel adjacent to the Big Qualicum River on Vancouver Island, British Columbia.

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The facilities of the artificial channel made possible both an accurate enumeration of chum salmon (*Oncorhynchus keta*) spawners and virtually complete recovery of carcasses, thus enabling the determination of tag loss by measurement of changes in the tagged:untagged ratio during the spawning period. In addition to the basic measurement of tag loss, experiments were conducted to test the effect of disk size on rate of loss and the effectiveness of accessory buffer disks in reducing loss.

Materials and Methods

Chum salmon migrate into the river in a fully mature state after a period of delay off the river mouth (Walker and Lister MS, 1965). The migration takes place over a five-week period with peak spawning occurring in the last week of November and the first week of December. The average life span in the river, approximately 10-11 days, is highly dependent on the stage of the run (Fig. 1). On the basis of observations of distinctively tagged female chum salmon, the average spawning ground life span is comprised of 7 days of spawning activity and 3 days of holding prior to movement onto riffle areas.

Chum salmon were counted into the 2000-foot long channel through a removable panel in a barrier fence at the downstream end. A diffuser system covered with a grating prevented fish from leaving the channel at the upstream end. Tagged fish were released into the channel from a trap located at the entrance. Tags were applied throughout the migration period each year but it was not practical to tag a constant percentage of the run. Carcasses were recovered daily from both the channel entrance fence and the 6 five-foot-deep holding pools which are distributed over the length of the channel.

Ward (1959) observed that as many as 10 percent of the tags available to recovery in a particular area may be overlooked during the first examination of carcasses. In 1965 and 1966 a total of 1105 carcasses recovered from the channel were re-examined to assess the incidence of overlooked tags. Only one tag was recovered on re-examination. It was calculated that of the 546 tags available for recovery in those years 4.3 (0.8%) were overlooked on the initial surveys. Because of the small error introduced from this source, no correction has been applied to tag loss calculations.

The standard tags used in the study were laminated cellulose acetate disks of various colours, $\frac{7}{8}$ in. in diameter and .049 in. thick[1]. The effect of disk size on rate of loss was tested by means of paired releases of males bearing the standard tag and a smaller disk ($\frac{9}{16}$ in. diam., .042 in. thick). A paired-release experiment was also conducted to assess the effect on tag loss of placing transparent buffer disks ($\frac{1}{4}$ in. diam., .049 in. thick) on the tag pin outside each standard large disk.

All tags were attached by a 3-in. nickel pin (.035 in. diam. shank)[2] inserted through the musculature below the anterior portion of the dorsal fin insertion. One of the two disks applied to each fish possessed an identifying number. Particular attention was given to making sure that tags were not applied too snugly, in a manner that would result in continual strain on the disks. Fish were not anesthetized prior to tagging.

[1]Manufactured by Howitt Plastics Co., Molalla, Oregon, U.S.A.

[2]Manufactured by Noesting Pin Ticket Co., New York, N.Y., U.S.A.

Tag loss has been estimated by the following formula:

$$\text{Percent tag loss} = \frac{\left(\begin{matrix}\text{Estimated total population}\\ \text{recovery rate}\end{matrix}\right) - \left(\begin{matrix}\text{Recovery rate}\\ \text{of tagged group}\end{matrix}\right)}{(\text{Estimated total population recovery rate})} \times 100$$

Virtually 100 percent of the population counted into the channel was recovered each year (Table I). Yearly estimates of actual recovery rates shown in Table I were used to calculate yearly tag loss rates for each sex and group of tags.

Table I.

Numbers of chum salmon entering the spawning channel and percent recovered

Year	Number Counted into Channel	Number of Carcasses Recovered			Percent Recovered
		Male	Female	Combined Total	
1963	2308	988	1303	2291	99.26
1965	2207	1166	999	2165	98.10
1966	2608	1198	1409	2607	99.96

The relationship between spawning-ground life-span, date of entry into the channel and rate of tag loss was examined by correlation and multiple regression analysis (linear model). The null hypothesis was tested in each case by Student's t-test. The estimate of tag loss for a given tagging date may represent from 8 to 21 recoveries (mean 15) in the case of males and from 13 to 43 recoveries (mean 26) in the case of females. Regression calculations were not weighted to account for these sample size differences. Each sex was treated separately and all three years of data were used to calculate the respective correlation and partial regression coefficients.

The degree of egg retention in a random sample of tagged and untagged females was assessed to determine if the tagging process caused any pre-spawning mortality or lowered the chance of successful egg deposition. A small number of females which had lost tags may have been included in the untagged group but this would not have affected the results significantly.

Results

Effect of Tagging on Spawning Success

A comparison of egg retention levels in both tagged and untagged females during 1963 is presented in Table II. The results indicate that tagged fish were at least as successful in egg deposition as untagged fish. It is concluded that the tagging process did not impair the fish to any measurable extent.

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Tag Loss

Yearly estimates of percentage tag loss (Table III) show the existence of a significant difference between sexes: overall averages of 30.3 percent for males and 9.4 percent for females. The between years differences within each sex are remarkably small.

Table II.

Comparative spawning success of tagged and untagged female chum salmon in 1963

	Tagged	Untagged
Sample size	58	425
Partially spawned[a]		
Number	6	68
Percent	10.3	16.0
Unspawned[b]		
Number	3	28
Percent	5.2	6.6

[a]Egg retention ranging from 100 eggs to 50% of estimated fecundity.

[b]Egg retention greater than 50% of estimated fecundity.

Table III.

Tag loss estimates by sex and year of tagging

Sex	Year	Number Tagged	Number Recovered	Percent Loss
Male[a]	1965	101	70	29.4
	1966	93	64	31.1
Female	1963	140	126	9.3
	1965	144	127	10.1
	1966	130	119	8.4

[a]1963 data excluded from comparison because males were only tagged during second half of run.

Tag loss among males was significantly correlated with both spawning-ground life-span ($r = +.63$; $P = <.05$) and progression in the date of entry ($r = -.79$; $P = <.01$) (the latter relationship shown in Fig. 2). Because of the strong association between life-span and entry-date (Fig. 1) partial regression coefficients were calculated. The partial regression of tag loss on entry-date did approach the 5 percent significance

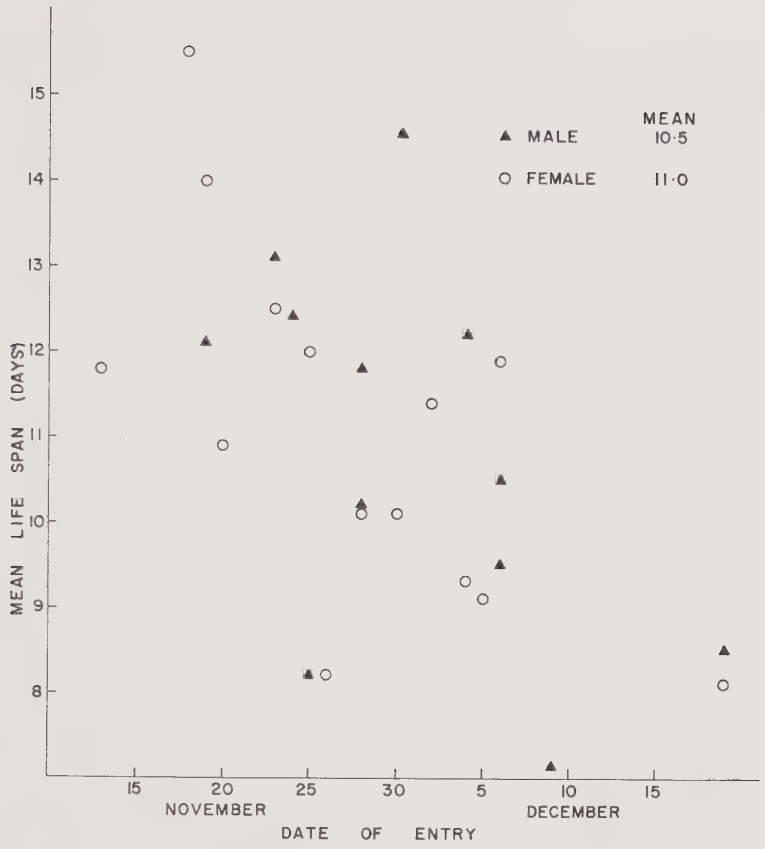

FIGURE 1. *Spawning ground life span of chum salmon in relation to date of entry into the spawning channel.*

level (t = 3.78; P = .07), but regression of tag loss on life-span independent of entry-date dropped sharply in significance (t = 2.45; P = >.5), thus indicating that a high correlation with tag loss resulted largely from a correlation with entry-date.

Tag loss among females was less variable than among males (Fig. 2) and was not significantly correlated with either date of entry (r = .57; P = <.1) or spawning-ground life-span (r = .27; P = >.1).

Reduction of Tag Loss

The placement of a buffer disk (¼ in. diam.) on the pin outside each large disk ($\frac{7}{8}$ in. diam.) reduced tag loss by approximately 50 percent (Table IV). The number of recoveries in the test group was significantly higher than that in the control group (chi-square = 13.7; P = < .005).

In the 1966 experiment loss of $\frac{9}{16}$ in. diameter disks was lower than that of the standard size disks, but the difference in number of recoveries from each group was not highly significant (chi-square = 3.5; P = <.1). The results do suggest that the greater thickness (.049 in. versus .042 in.) of the larger disk is not an advantage.

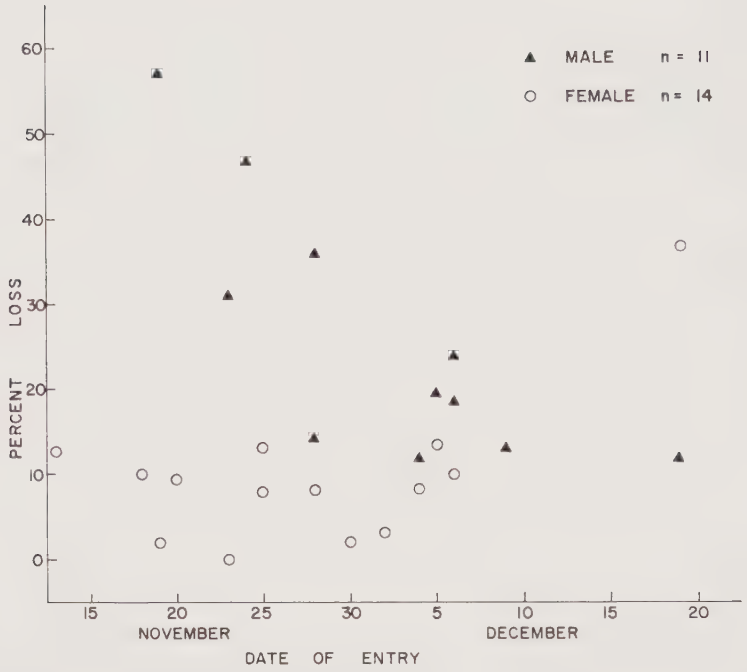

FIGURE 2. *Rate of tag loss in relation to date of entry into the spawning channel.*

Table IV.

Comparative loss of three tag types as indicated from paired releases of tagged male chum salmon

Year	Disk Tag Description	Number Tagged	Number Recovered	Estimated Percent Loss
1965	7/8 in. diam.	76	49	34.3
	7/8 in. diam. with 1/4 in. diam. buffer disks	76	62	16.8
1966	7/8 in. diam.	93	64	31.2
	9/16 in. diam.	93	72	22.6

The above chi-square values were calculated with an adjustment for continuity (Snedecor, 1956, p. 217).

Discussion

The rates of loss measured in this study would, assuming even sex ratio, result in a Petersen-type population estimate (Ricker, 1958) with a positive bias of 27 percent. Bias in the male and female estimates would amount to 43 percent and 10 percent respectively. The magnitude of error corresponds closely with that determined from estimates of pink salmon (*Oncorhynchus gorbuscha*) escapements to the Lakelse

River, British Columbia, in which tag and recovery estimates by the Fisheries Research Board of Canada over a three-year period averaged 30 percent higher than the fence count (H. D. Smith, unpublished manuscript).

Vernon et al. (1964) reported a similar difference between sexes in degree of bias for adult pink salmon tagged on migration through a counting fence. The subsequent male and female population estimates were higher than the actual count by 25 percent and 11 percent respectively. In the same experiment, tagging of pink salmon at the stream mouth and in the river downstream of the counting fence produced larger errors, 42 percent and 34 percent respectively, than tagging of more mature fish at the counting fence; but in both cases, however, the errors for each sex were nearly equal, i.e., males 33 percent, females 36 percent. This would suggest that prior to the actual spawning phase other types of loss among females may compensate for the higher rate of spawning ground loss among males.

The consistent difference between sexes in rates of tag loss is probably due to behavioural differences. Mathisen (1962) observed that spawning male sockeye salmon (*Oncorhynchus nerka*) tended to migrate over a much larger area than females, and Fabricius and Gustafson (1954) noted that spawning male arctic char (*Salvelinus alpinus*) showed successive polygamy. These observations are applicable to salmonids in general, and as a result it might be expected that the male would normally experience a greater number of aggressive encounters than the female, and that this would favour a greater rate of tag loss from males.

Approximately 4 percent of the tags recovered in this study consisted of only one disk. Once a disk is missing there would appear to be a high probability of shedding the tagging pin and remaining disk. The effectiveness of buffer disks in reducing loss indicates that disk failure was the primary factor responsible for tag loss.

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The Application of an Air-Percolation System for Water Temperature Reduction in Robertson Creek

by

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Abstract

High-water temperatures have been a principal factor in limiting the success of the Robertson Creek Spawning Channel. To alleviate this problem two air percolation systems were used. The upwelling of cold water induced by these systems significantly altered the temperature profiles of the head waters, Boot Lagoon and Great Central Lake. The surface water temperature of Boot Lagoon was reduced 6.5 degrees F. after three days of percolation.

Concomitantly, the water temperature in Robertson Creek dropped 6.0 degrees F. One-half of this decline was probably caused by percolation, while the remainder was due to a drop in air temperature. Inflowing warm water from Great Central Lake at the time of the experiment reduced the extent of temperature reduction in Boot Lagoon and Robertson Creek.

Introduction

The Robertson Creek Spawning Channel, a secondary outlet of Great Central Lake, is located on the west coast of Vancouver Island about 100 miles west of Vancouver, British Columbia (Fig. 1). This spawning channel (Lucas 1960) was established in 1959 for the examination of salmon development and transplantation techniques. Controlled water volume, depth, velocity and substrate are appropriate for the intended purposes; however, several environmental factors interfere with the successful operation of the channel. The main problem is the adverse effect of high-water temperatures on maturing and spawning adult salmon.

The upper lethal temperature limits of salmonids have been studied by physiologists (Black, 1953; Brett, 1952; and others). The lethal limit for salmonids is about 75.0°F. Black (1953) found that the median tolerance limit ($L.D._{50}$) of *Salmo gairdnerii* was 75.2°F while the temperature causing 100 percent mortality ($L.D._{100}$) was

78.0°F—the stress zone began at 72.0°F. Robertson Creek water temperatures during salmon migrations and spawning have ranged from 68.0°F to 73.0°F. These temperatures probably approach the stress zone for salmon.

As an example, high-water temperatures during the late Stuart River run caused an increased rate of mortality among unspawned sockeye (Andrew and Geen, 1961). Wickett (1962) notes that higher water temperature during incubation may be the reason that pink salmon do not occur in the Puntledge River to the same extent as in the Morrison and Tsolum rivers.

These higher temperatures adversely affect salmonid survival, but they may be near optimum for infectious diseases such as furunculosis, columnaris and such protozoan diseases as trichodiniasis and *Ichthyophthirius multifilis* (Wood, 1965). Concentrations of the endemic parasitic fungus *Saprolegnia declina* are directly proportional to water temperature (McKay, 1967).

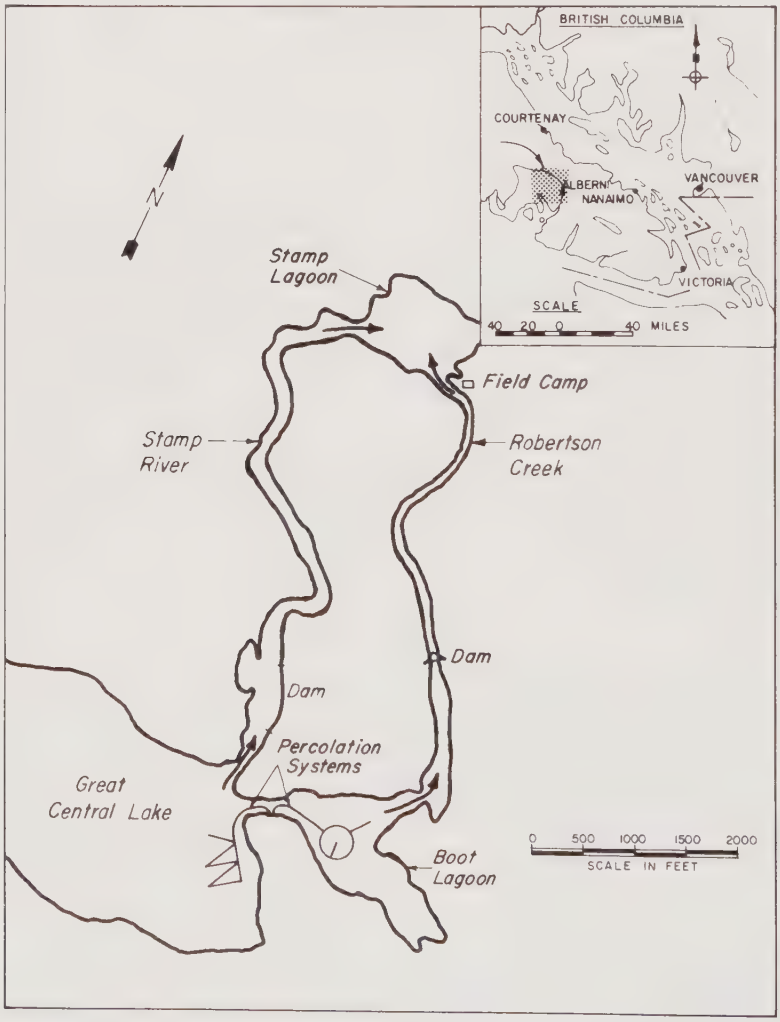

FIGURE 1. *Showing the location of the Robertson Creek spawning channel.*

Epidemic columnaris has appeared suddenly when temperatures exceeded 55.0 to 56.0°F in closely grouped migrating and spawning sockeye salmon (Wood, 1965). Wood recommends specifically that water temperatures on the spawning grounds be maintained at 54.0°F or below if possible. High-water temperature and associated disease organisms is probably one important cause for the significant pre-spawning mortality of adult salmon and their progeny during the rearing period in Robertson Creek. Another benefit of the cooler water may be to induce migration of adults into the spawning channel from the Stamp River Lagoon (Fig. 1).

The compressed-air bubbler system induces water circulation in lakes. This technique creates upwelling from the lower strata to the lake surface. If used during the summer stagnation period, it lifts and mixes cold, dense water (about 39.0 to 40.0°F) with the warmer surface water. Great Central Lake and Boot Lagoon have a reservoir of cold water which can be used for reducing water temperatures in Robertson Creek.

Methods and Materials

A limnological survey was conducted at the outlet area of Great Central Lake and in Boot Lagoon in early August 1966. Depth soundings were taken with a Furuno F701 recording echo sounder. Oxygen determinations were made by means of the unmodified Winkler method, and vertical temperature series were taken with a Telethermometer (Y.S.I., model 44TH). Water movement in the channel connecting Great Central Lake and Boot Lagoon was measured with a Gurley Current Meter.

A bubbler system, described by Halsey (1968) (2800 ft. circumference) was installed with two 225 ft. spurs (Fig. 2). The Great Central Lake system was installed with a different configuration at greater depth (Fig. 2), and with a total length of 2400 ft. Each system was connected to an Atlas Copco VT4, 160 ft^3/min compressor by a 600 ft feeder line (100 psi). The compressors operated at a pressure of 100 psi.

The bubbler system operated in Boot Lagoon from August 25 to 31 (144 hrs) and from September 20 to 29 (240 hrs); in Great Central Lake it operated from September 9 to 29 (504 hrs).

Results

Great Central Lake has ample depth for the operation of a bubbler system. On the south side of the channel entrance into Boot Lagoon a relatively deep area provides an ample reservoir of cold water (Fig. 2, Fig. 4, September 4).

The draw off from Great Central Lake by Stamp River and Robertson Creek assures a movement of lake water toward these outlets. Therefore any cold water brought to the surface by percolation in Great Central Lake would likely be available to the Boot Lagoon entrance and subsequently to Robertson Creek. The temperature profile of August 12 is typical of a thermally stratified lake (Fig. 3). Artificial circulation was started on August 25 and the stratification on that date was most likely the same as that of August 12. The reduction of the epilimnion temperature from 68.0 to 61.5°F and the gradual warming of the hypolimnion between August 25 and 27 is a

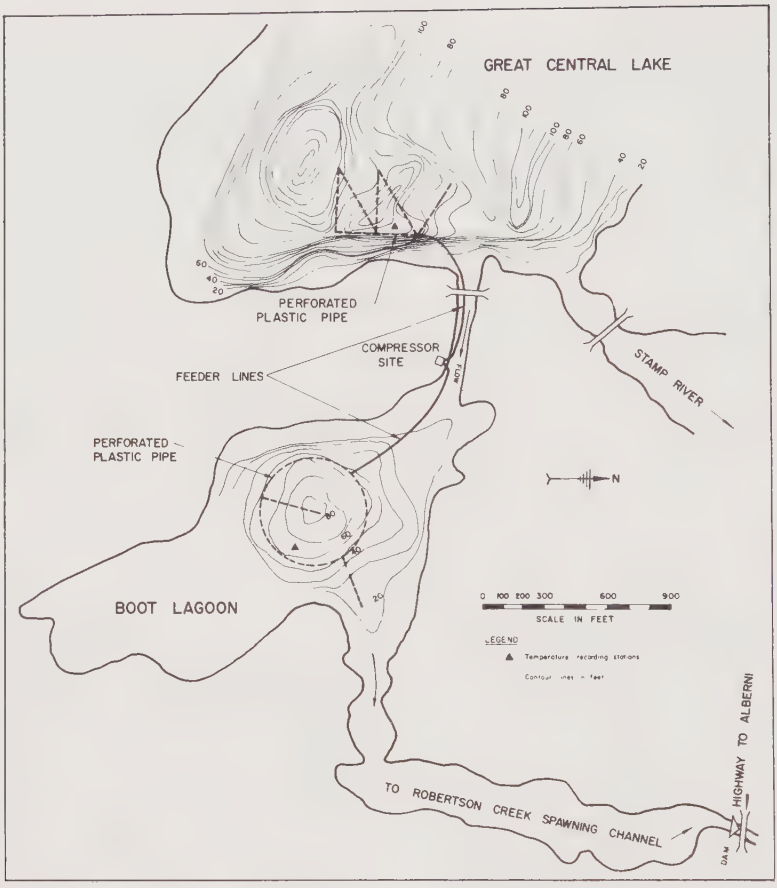

FIGURE 2. *Air percolation systems in the head waters of Robertson Creek (Boot Lagoon and Great Central Lake).*

change that is typical of artificial circulation (Schmitz and Hasler, 1958). A nearly isothermal condition was reached by August 31 at about 62.0°F (Fig. 3). The subsequent warming of the surface water was partly due to the inflow of warmer water from the surface of Great Central Lake.

A shortage of oxygen in the lower hypolimnion of Boot Lagoon was noted prior to percolation. Analysis of dissolved oxygen was made at several locations including Robertson Creek before and during percolation. No changes in dissolved oxygen were evident in Robertson Creek during the operation of the percolation system.

Great Central Lake's temperature profile (Fig. 4, September 4) illustrates a thermal stratification comparable to that of Boot Lagoon prior to percolation. Subsequent percolation had the same cooling effect on the epilimnion of Great Central Lake as that recorded for Boot Lagoon.

The atypical temperature series recorded in the hypolimnion of Great Central Lake may well be accounted for by seiche action which is not uncommon in this

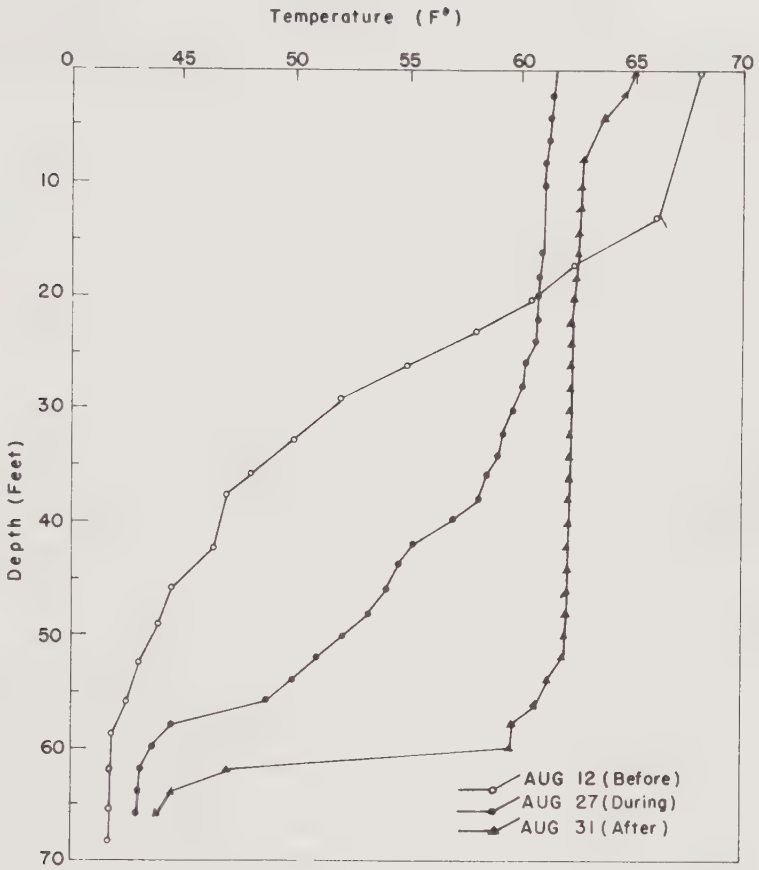

FIGURE 3. *Boot Lagoon temperature profiles before, during and after artificial circulation in August, 1966.*

lake (F. Boyd, Pers. Comm.). Other indications of seiche action were observed in the channel connecting Boot Lagoon and Great Central Lake. Water movement measured by a current meter indicated that there was an occasional periodic outflow of water from Boot Lagoon to Great Central Lake (opposite to the direction of natural flow). This evidence further suggests seiche action as the probable cause of the reversing water movement.

In order to establish the relationship that exists between daily mean air and water temperature, a linear regression analysis was performed. Seven day periods from July 1 to August 24 were compared with the percolation period, August 25 to 31. All other percolation periods (Sept. 20–29, Sept. 9–29) as well as the Sept. 1–8 interval were analyzed in the same manner.

The irregular variability of the air temperature precluded linear regression. Therefore the statistical analysis did not demonstrate a causal relationship between air and water temperature. The data are therefore assessed empirically.

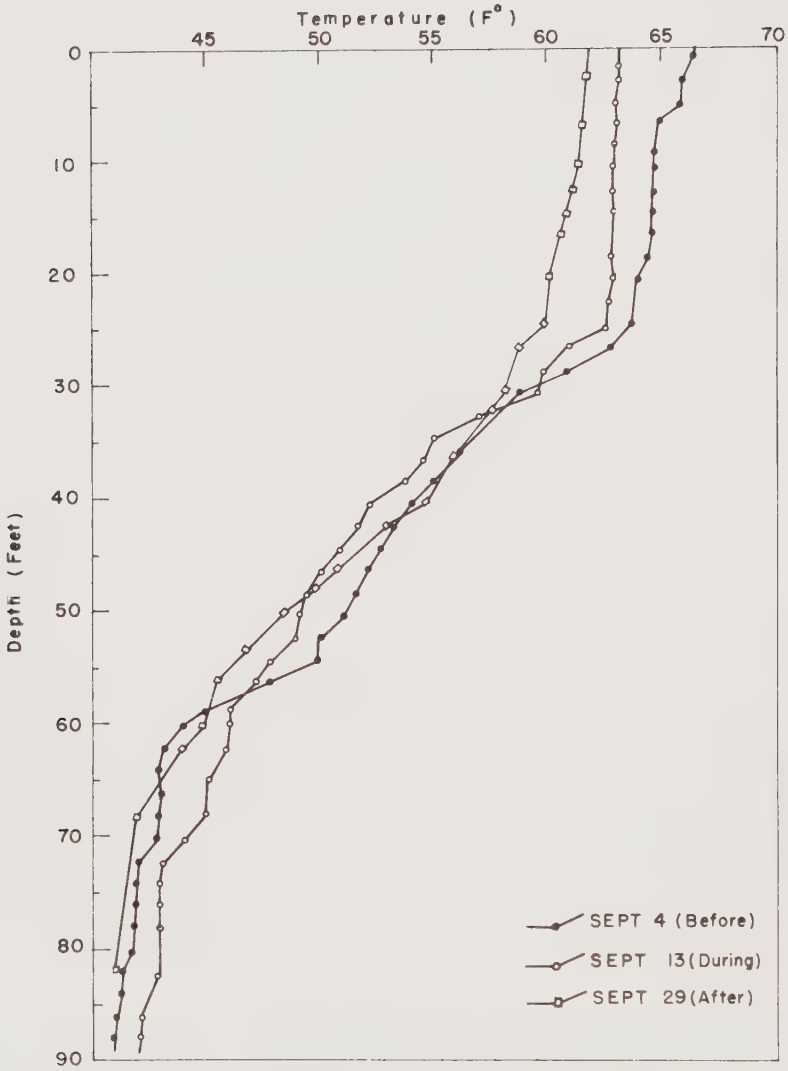

FIGURE 4. *Great Central Lake temperature profiles, before, during and after artificial circulation, September, 1966.*

Discussion

McCombie (1959) studied the relationship between air and water temperature in South Bay, Lake Huron, and found that a close correlation existed between the monthly means of lake surface water and air temperature. The relation between these two quantities was described as a linear regression.

The 6.0°F decrease in water temperature of Robertson Creek during the August 25–31 percolation may not have been only a response to the decline in air temperature (Fig. 5). The response of mean water temperature to previous declines in air temperature suggest that the alterations were not entirely accounted for by air temperature

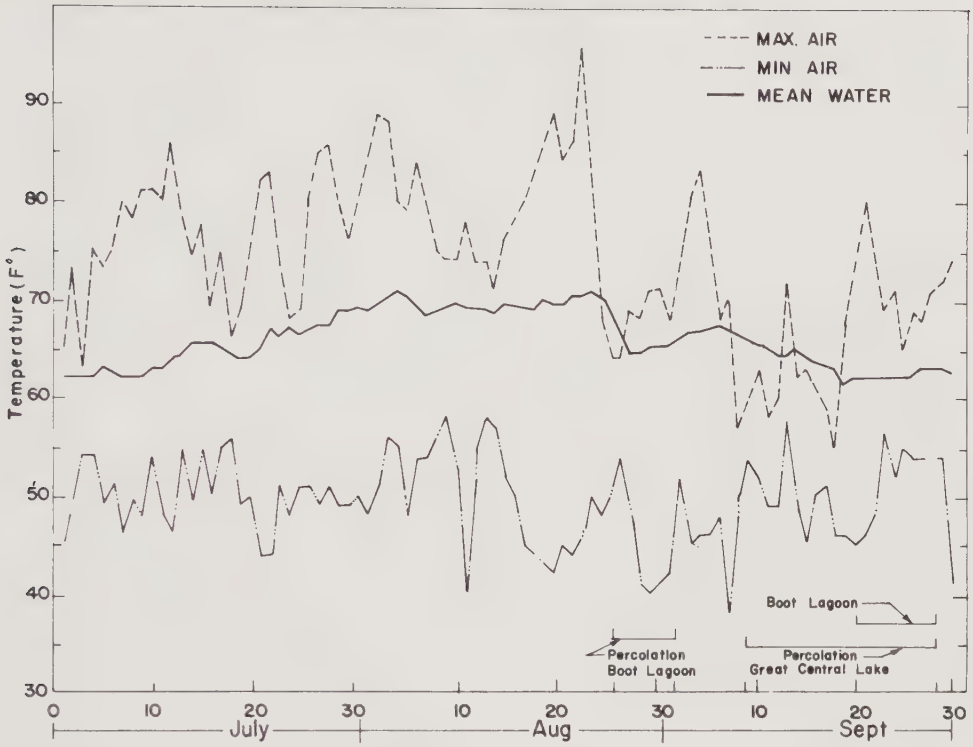

FIGURE 5. *Robertson Creek temperature regime from July to September, 1966. Mean water temperature was recorded at the sub-channel station.*

changes. A relationship between these two factors is suggested by the temperatures recorded before and during percolation at Robertson Creek (Table I).

The maximum air temperature for the period August 2 to 14 declined 18.0° from 88.0 to 71.0°F, and for the same period the minimum increased 2.0° fluctuating between 58.0 and 40.0°F. However, over the same period the mean water temperature dropped only 2.0°. Similarly, within the percolation period of Boot Lagoon, the

Table I.

Air and water temperatures at Robertson Creek before and during percolation.

Time Period	Change in Maximum Air Temperature (degrees F)	Change in Minimum Air Temperature (degrees F)	Change in Mean Water Temperature (degrees F)
Aug 2–14 (before)	−18.0	+2.0	−2.0
Aug 23–27 (during)	−32.0	+8.0	−6.0

maximum air temperature dropped 32.0° (96.0–64.0°F) from August 23 to 27 and significantly the minimum increased 8.0°, while at the same time the mean water temperature dropped 6.0°.

Because there was a 2.0° drop in water temperature for net 16.0° decline in air temperature (August 2–14), the net 24.0° decrease in air temperature (August 23–27) therefore should cause approximately a 3.0° drop in water temperature. However, there was a 6.0° decline. The remainder of the decrease (3.0°) was probably the result of percolation in Boot Lagoon. Furthermore, had there not been an inflow of warm water from the surface of Great Central Lake, the decrease in water temperature would likely have been greater. When the percolation of Boot Lagoon was stopped (August 31), the mean water temperature in Robertson Creek rose 2.5° while the maximum air temperature increased 20.0°. This relationship between water and air temperature change is approximately the same as that of the August 23–27 period: a 3.0°F water temperature response to a 32.0° air temperature change.

The percolation of Boot Lagoon was stopped on August 31 to preserve the remaining hypolimnion. The second percolation system was started on September 9 in Great Central Lake to provide cooler water to Boot Lagoon. However, the percolation system was prematurely terminated after 504 hours (September 29). Had percolation continued beyond that date, it is probable that the epilimnion of the percolation area would have been cooled further, and, as a result, have lowered the temperature in Boot Lagoon and subsequently in Robertson Creek.

From September 18 to 20 the maximum air temperature increased from 55.0 to 74.0°F, yet the surface water at the Great Gentral Lake percolation site remained constant at 60.5°F. This was likely due to the effects of the percolation system.

The results of the percolation project clearly revealed two facts: first, that temperature regimes can be substantially altered with ease, and second, that for significant results percolation must take place for a suitable period of time. Though the work reported here did not operate the precolation system long enough for various reasons, the inconclusiveness of the results does not invalidate the usefulness of the technique.

Summary

1. Percolation of Boot Lagoon and Great Central Lake significantly altered their water temperature profiles.
2. Part of the temperature decline recorded in Robertson Creek was attributed to percolation.
3. Sufficient percolation could possibly reduce the incidence of disease and temperature stress of salmon in Robertson Creek.

Acknowledgments

The authors would express their appreciation to Mr. Dixon MacKinnon, Chief Biologist of the Department of Fisheries, and to Mr. E. H. Vernon, Chief Fisheries Biologist of the Fish and Wildlife Branch, for their interest and support in this project. Mr. R. K. Kearns, Robertson Creek project biologist, kindly lent his advice and assistance. The cooperation and valuable assistance of Mr. J. B. Baker, resident technician, and his field staff, were very much appreciated.

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A Converging Throat Trap for Sampling Juvenile Salmonids

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Abstract

A converging throat trapping system is described which provides a sample of downstream migrating juvenile salmon. The traps have been operated from a counting fence to enumerate newly emerged sockeye salmon fry. They have fished successfully in depths to four feet and velocities up to six feet per second with no observed mortalities to the captured fish.

Introduction

Each spring for the past six years estimates have been made of the sockeye fry production from Fulton River, one of the principal spawning streams of the Babine Lake System in north central British Columbia. These estimates, which ranged from 21 to 46 million over the six year period, were based on a systematic sampling of the downstream migrating fry by a vertical ladder-type trap. The trap consisted of a vertical ladder with a number of bag nets attached. The entire assembly was operated from a double-pontoon raft.

It has been difficult, however, to standardize the trapping procedure for two principal reasons. First, the period of downstream migration of the fry coincides with spring runoff, and discharges from 50 to 6,080 cfs have been metered. It has not been physically possible to operate the trapping gear on a continuous basis at the maximum discharge. Second, debris collected on the screening surfaces and rapidly changed the efficiency of the gear. A further disadvantage of the sampling gear was the excessive mortality suffered by the captured fry. To overcome these problems a counting fence was constructed near the mouth of the Fulton River and a trapping apparatus used by Tait and Kirkwood (1962) for pink salmon fry enumeration at Prince William Sound, Alaska, was modified and employed on the fence during the 1967 program (Fig. 1). The fence is equipped to count both the adult and fry migrations; however, this report will describe in detail only the downstream trapping facilities.

Description of the Trap

A unit consists of a rectangular tunnel and an attached oval retainer (Fig. 2). The apparatus, constructed of a welded aluminum frame, is covered on the sides and floor with 0.028″ diameter galvanized steel screen. The tunnel converges from a one-foot wide entrance to a $1\frac{3}{4}$″ wide exit (or throat) which is continued inside

FIGURE 1. *Fulton River counting fence.*

the oval retainer. A slider measuring $8' \times 1\frac{3}{4}'' \times \frac{1}{4}''$ fits into the throat for the emptying operation. The tunnel has been tested at 20 cfs and is designed for a maximum of 30 cubic feet per second. The oval retainer measures $10'' \times 12''$.

The walls, starting at the throat and continuing around 6/10 of the circumference of the retainer, are of sheet aluminum (Fig. 3). This sheeting is carried 8″ into the retainer on each side of the throat to provide for low velocity resting-areas. The design thus minimizes mortality of captured fry within the retainer (Fig. 4). As noted below, this aluminum sheeting also facilitates emptying. The downstream end of the retainer is screened with the top open and the bottom plate hinged. The latter, equipped with a rubber seal, can be quickly opened to provide entry to the inner portion of the retainer for repair or maintenance.

Operation of Gear

In fishing position, the entire assembly is suspended from the fence and pivoted on an axle located at the top of the tunnel entrance (Fig. 5).

To empty the trap a slider is inserted in the throat and thrust to the floor. The entire trap assembly is then raised to slightly past the horizontal position (Fig. 6). The fish and debris in the retainer flow along the slider (confined in the throat area) and on the aluminum sheeting of the retainer. The retainer is cleaned by flushing the trap with water from a pressure hose or bucket. The entire contents fall onto a

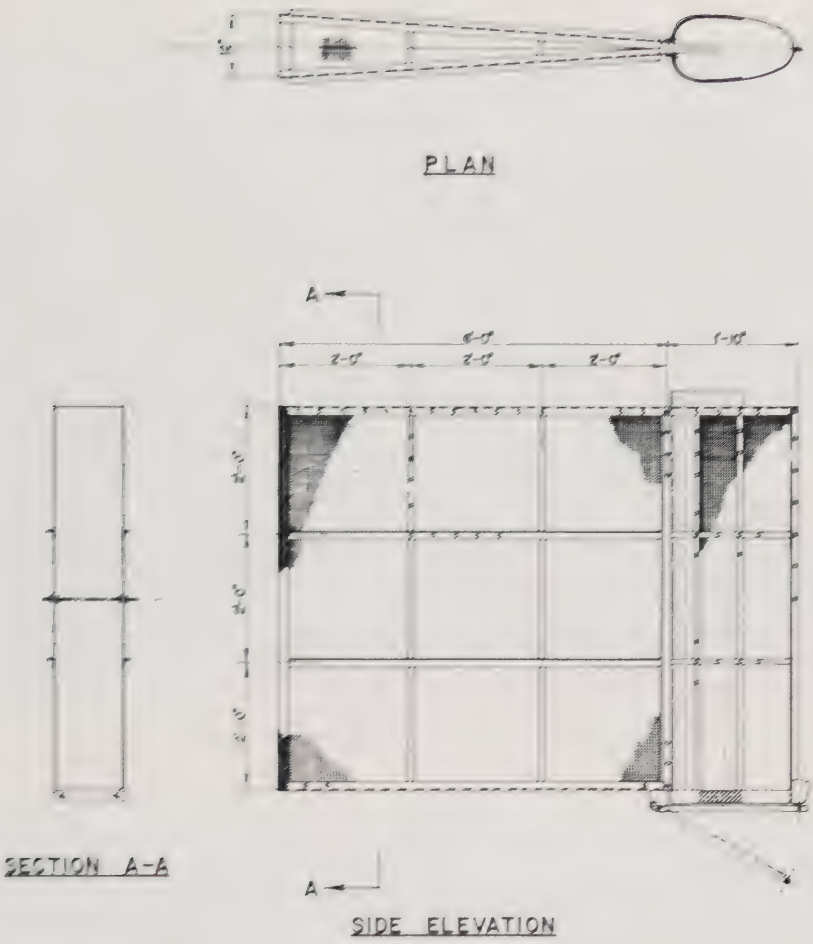

FIGURE 2. *Plan and elevation of converging throat apparatus.*

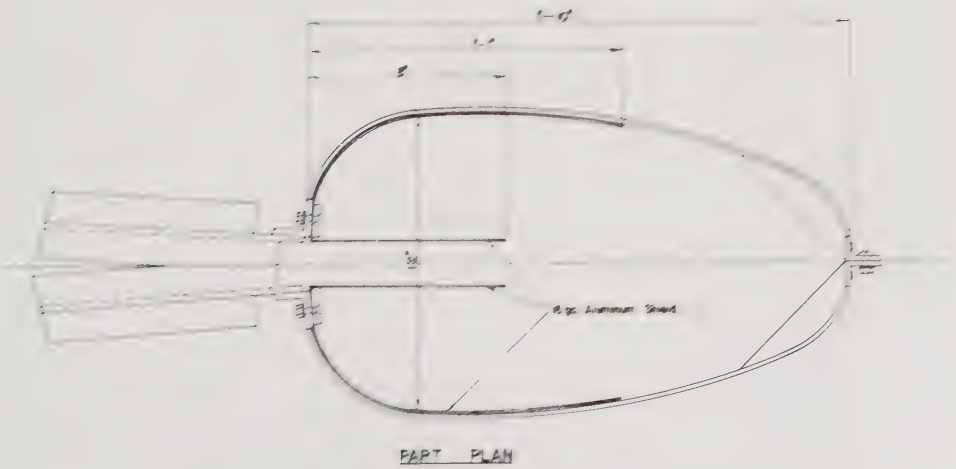

FIGURE 3. *Plan of oval retainer. Note extension of aluminum shield into retainer.*

FIGURE 4. *Oval retainer in fishing position. Note salmon juveniles on either side of the throat.*

FIGURE 5. *Apparatus in fishing position.*

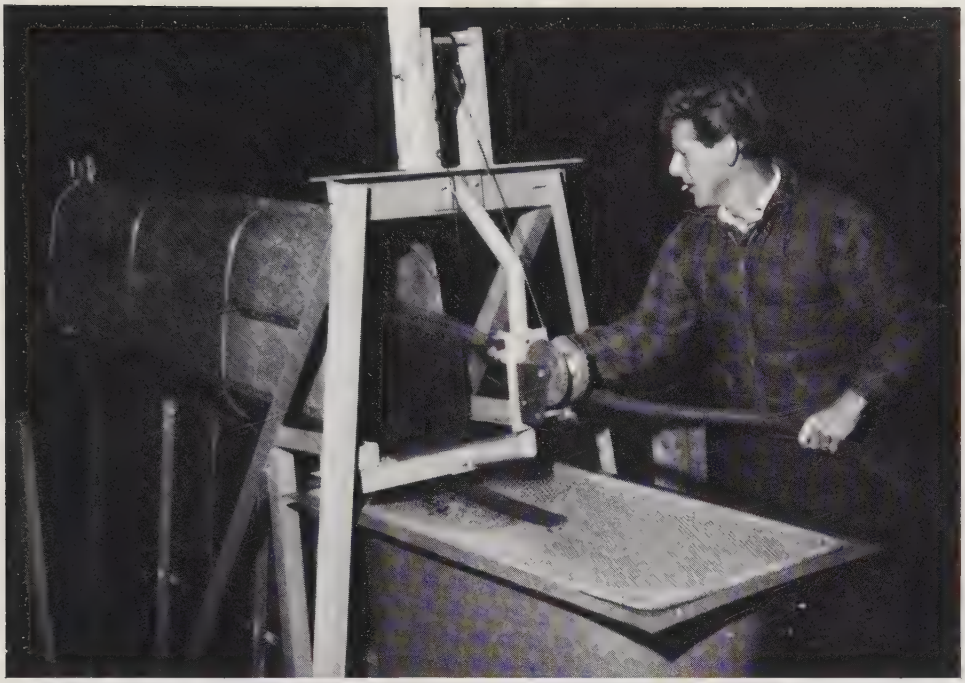

FIGURE 6. *Apparatus in emptying position.*

nylon mesh which retains coarse debris. The fry and finer material pass through the mesh into a container located below. The catch may be enumerated from the container or removed to a more suitable site.

The converging throat trap has been operated in depths of water ranging from 2 inches to over 4 feet and at average velocities of up to 6 feet per second. Visual comparison of velocities in the tunnel with those adjacent to the trap indicate it is quite capable of handling velocities of this magnitude.

Sampling Procedure

Twelve traps located 20 feet apart were installed. Each trap fished midpoint of the distance between the piling supporting the fence. Three simultaneously operated traps served as the basic sampling combination for a particular night, and differing combinations were applied on successive nights to more or less equalize the physical stresses on all gear and thus avoid breakdowns. In addition, the rotation of gear provided a secondary measure of the relative catching ability of each trap.

The fishing period started at 2200 hours and terminated no earlier than 0200 hours or no later than 0500 hours. Following a series of experiments, the standard fishing time was established at 20 minutes, with 5 minutes allowed for raising, emptying, cleaning and re-setting the gear. The trap therefore fished eighty percent of a given sampling period. The proportion of fish migrating in a fishing period in relation to that occurring in a chronological day was measured periodically by fishing for a full 24 hour period.

The relative efficiency of each trap was determined frequently by fishing all traps simultaneously close to or during the peak of night migration for at least two twenty-minute periods.

Remarks

Catches for a 20-minute fishing period have ranged as high as 4,392 sockeye salmon fry. Coho salmon (*Oncorhynchus kisutch*) yearlings have also been captured. No immediate mortality of fry or smolts has been observed. The indication from the first season of operation is that the converging throat trap largely eliminates problems which affect the efficiency of other types of gear and provides a reliable means for capturing fry under a large range of stream conditions.

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A Modified Wolf Trap for Downstream Migrant Young Fish Enumeration

by D. Brent Lister, Richard A. L. Harvey and C. E. Walker

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Abstract

An inclined screen trapping system is described which provides a sample of downstream fish migrations by screening a portion of the stream flow. The traps have been used to enumerate four species of juvenile migrant salmonids, ranging in size from emergent chum salmon fry (*Oncorhynchus keta*) to two-year-old steelhead trout smolts (*Salmo gairdneri*).

Introduction

The inclined screen trap, described by Wolf (1950), has been used extensively for enumeration of downstream migrating fish in relatively small streams. This type of trapping system is generally designed to screen the total stream flow; however, in applying the method to a relatively large stream it is advantageous to screen only a portion of the flow to reduce operational costs and to minimize any effects of handling on the fish population. A modification of the Wolf trap has been developed to sample juvenile salmonid migrations in the Big Qualicum River on Vancouver Island, British Columbia. The enumeration methods have been briefly described by Lister and Walker (1966).

Description

A systematic sample of migration is obtained with seven one-foot-wide inclined screen channels anchored to the base of an enumeration fence at equal intervals across 60 feet of stream (Fig. 1). Individual channels can lead either into an attached live-box, as shown in Figure 2, or into a common trough which in turn conducts fish into a series of live-boxes located at the side of the stream. The screening material on the bottom and sides of each channel is bolted to a welded steel frame and consists of flattened, expanded steel (mesh openings 0.28 in. $\times$ 0.16 in.). Flat steel welded at 30° angle from the vertical, is used for the leading edges of the frame. The entire unit is painted in order to prevent rust and to produce a smooth screening surface.

The live-box frame has been constructed of either ¾ in. marine grade plywood or 0.16 in. thick plate aluminum. A sliding baffle, extending to within 4 in. of the floor, is located near the entrance to the box. This baffle reduces turbulence and the sliding feature allows large fish such as adult steelhead trout sufficient space to pass under it into the main holding area. A sloping floor (15°) and sliding plywood back have been incorporated to facilitate the removal of fish by draining. The emptying procedure

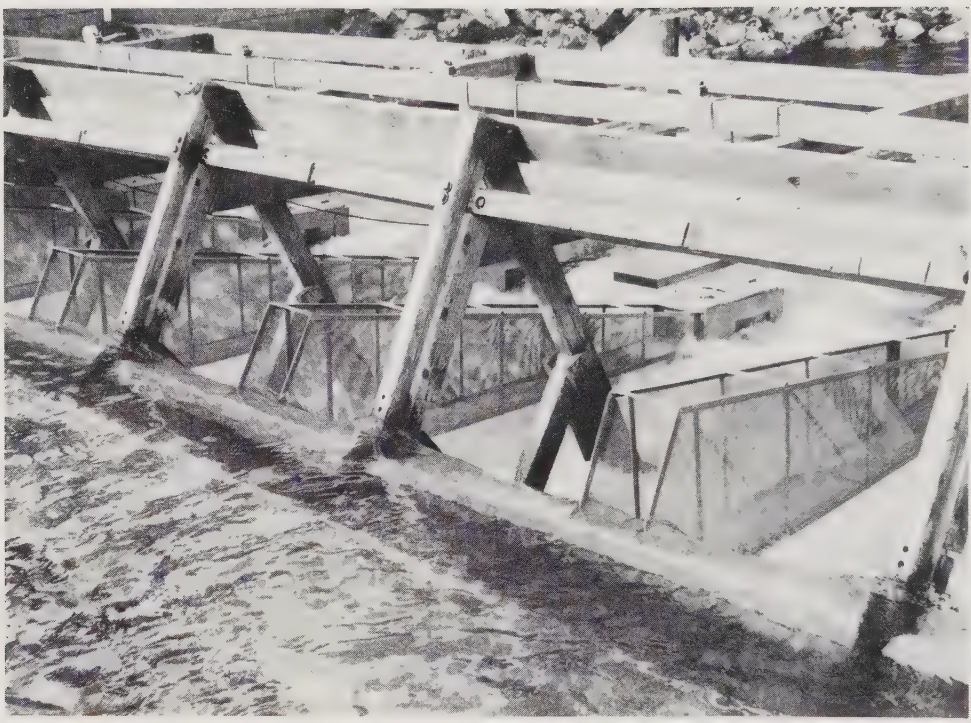

FIGURE 1. *Inclined screen traps anchored to the enumeration fence.*

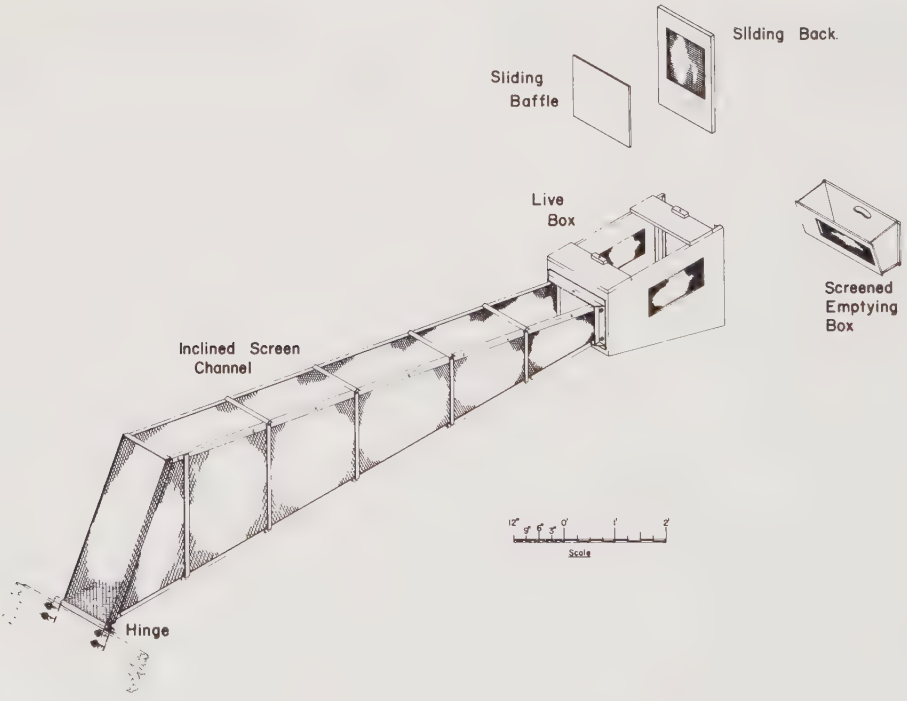

FIGURE 2. *Schematic drawing of the downstream migrant trap.*

involves raising the live-box well above water surface with a hand-operated winch and sliding the back of the box upward slightly to permit water, fish and debris to fall into a portable, screened container held at the lower edge of the live-box (Fig. 3). This method is faster and results in fewer injuries to the fish than dip-netting.

FIGURE 3. *Live-box emptying procedure. The screened side of the emptying box permits rapid separation of water from fish and debris.*

The mouth of each channel is anchored with a hinge to a 2 in. × 10 in. board which is supported on the fence base between A-frames (Fig. 2). A second 10 in board is placed on the supporting board to produce a drop into the channel. Water entering the channel accelerates to a velocity of approximately 5 feet per second between the

leading edges (Fig. 1). The trapping system is normally operated with vertical steel rods anchored at one-foot intervals on either side of each channel mouth (not shown in Fig. 1) in order to produce a turbulence effect comparable to that produced by the channel edges and thus possibly reduce the chance of gear avoidance.

The gear has been used over an eight-year period to enumerate juvenile salmonids ranging in size from chum salmon fry (*Oncorhynchus keta*), averaging 0.4 g. in weight, to steelhead trout smolts (*Salmo gairdneri*), with an average weight of 50 g. It has also captured substantial numbers of steelhead trout kelts. The average percent recovery of dye-marked emergent chum salmon fry released upstream of the traps (12.5%) has corresponded closely with the theoretical efficiency of the gear (13.5%) based on the percentage of stream discharge strained. Depending to some degree on approach velocity which becomes critical at low stream discharge, efficiency of the traps in capturing coho salmon (*Oncorhynchus kisutch*) smolts and steelhead trout smolts is variable. Population estimates for these species are based on mark and recapture.

An individual one-foot-wide channel efficiently screens a discharge of up to 8 cubic feet per second. The hinged attachment of the mouth makes it possible to adjust the angle of the screening surface to the flow in order to accommodate changes in stream discharge. Although the trapping system has been operated over a discharge range of 50 to 500 cfs it is best suited to relatively stable situations.

Acknowledgments

The authors are grateful to Stuart D. Phillips for his contribution over several years to various refinements of the gear.

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Effects of DDT Larviciding on Aquatic Fauna of Bobby's Brook, Labrador

by C. T. Hatfield*

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Abstract

Effects of a DDT larviciding program on aquatic fauna in a tributary stream of the Alexis River, Labrador, were determined.

Effects on fish were measured by holding trout (*Salvelinus fontinalis*) in cages located at six stations along the brook. Bottom invertebrate counts were taken every third day in plots at five of these stations. Drifting insects in the stream were sampled using one foot square fyke nets before, during and after DDT larviciding near each station. Water, mud and fish samples were collected for DDT content analysis by gas chromatography. One untreated station was used as a control. Water was also sampled for general chemistry.

No significant mortalities occurred in the caged fish, but many dead wild fish were collected with DDT content readings of their tissue ten times those of fish caught before larviciding began. DDT content of wild fish caught alive increased with time.

Bottom invertebrates were sharply reduced as a result of DDT larviciding at all treated stations. Caddisfly larvae were most affected. Drifting insect larvae in the stream increased exponentially during each DDT application.

General chemistry of water in the brook was unchanged by larviciding.

Introduction

A study was conducted on a DDT larviciding program to determine any effects on the aquatic fauna in a tributary of the Alexis River near Port Hope Simpson, Labrador. The literature contains many papers describing effects of aerial DDT sprays on aquatic life but no study had ever been done in Canada on this ground larviciding technique. Data were needed to determine if the method was successful in controlling the target organism without harming other aquatic fauna in the area.

Larviciding Method

Larviciding, developed by Hocking, Twinn and McDuffie in 1949, involves discharging from metering bottles a 10 percent DDT-fuel oil solution into a stream over a fifteen minute period. The amount of DDT used is calculated from a formula which gives a concentration of roughly 0.1 part per million DDT in the water over the 15 minute period.

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Repeat dosage times are determined by weekly checks for black fly larvae (*Simulium venustum*) on plastic cones submerged in the water. Application points on the treated brook were located at the head of tributaries and wherever the stream re-emerged from a lake.

Methods and Materials

Location of Sampling Stations

Biological and chemical sampling was conducted at five stations set up along a twelve mile section (1 mile = 1.6 km) of Bobby's Brook (Fig. 1). A sixth station, the control, was established on Western Brook.

Bottom Fauna Sampling

Bottom insect larvae were quantitatively sampled from 12 foot square plots (1 ft. = .305 m) laid out in fast moving water, 1-2 feet deep, at four treated stations and the control. Counts were taken every third day from 20 stones, randomly chosen from a plot and carefully returned after examination to their original resting places (Cope, 1961).

Drifting Insect Sampling

Drifting insect larvae numbers in the stream were measured by trailing a unit of three 12″ × 12″ fyke nets (1/24″ × 1/30″ mesh) in the current at each station for 15 minute sampling periods, before, during and after each DDT application.

Caged Fish

Direct effects of DDT on the brook trout, *Salvelinus fontinalis* (Mitchill, 1815), were determined by holding five angled fish for twenty-nine days in each of three cages spaced across the brook at every station. Checks for mortality were made every twenty-four hours and upon completion of the experiment, the fish were analyzed for DDT content.

Wild Fish

Wild trout were angled at each station before and after larviciding and analyzed for DDT content. Wild trout behaviour was recorded during the study.

Netted Fish

Other effects on trout feeding habits were investigated by gill netting fish at Station 3, before larviciding began, and three weeks after it ended. Lengths, weights and stomach contents of the fish were recorded.

Chemical and Physical Analyses

Dissolved oxygen and total alkalinity followed Am. Standard Methods (1965). Phenylarsene oxide was substituted for sodium thiosulfate. Total hardness was determined by titration with EDTA and hydrogen ion concentration with a glass-electrode pH meter accurate to within 0.1 unit. Turbidity was measured using a U.S. Geological Survey turbidity rod (Welch, 1948). Water samples were taken by filling quart metal tins at each station before, during and after larviciding. Water, fish and sediment samples were analyzed for DDT and metabolite content by Dr. R. Duffy (St. Dunstan's University, P.E.I.) using gas chromatography (Duffy and O'Connell, 1968).

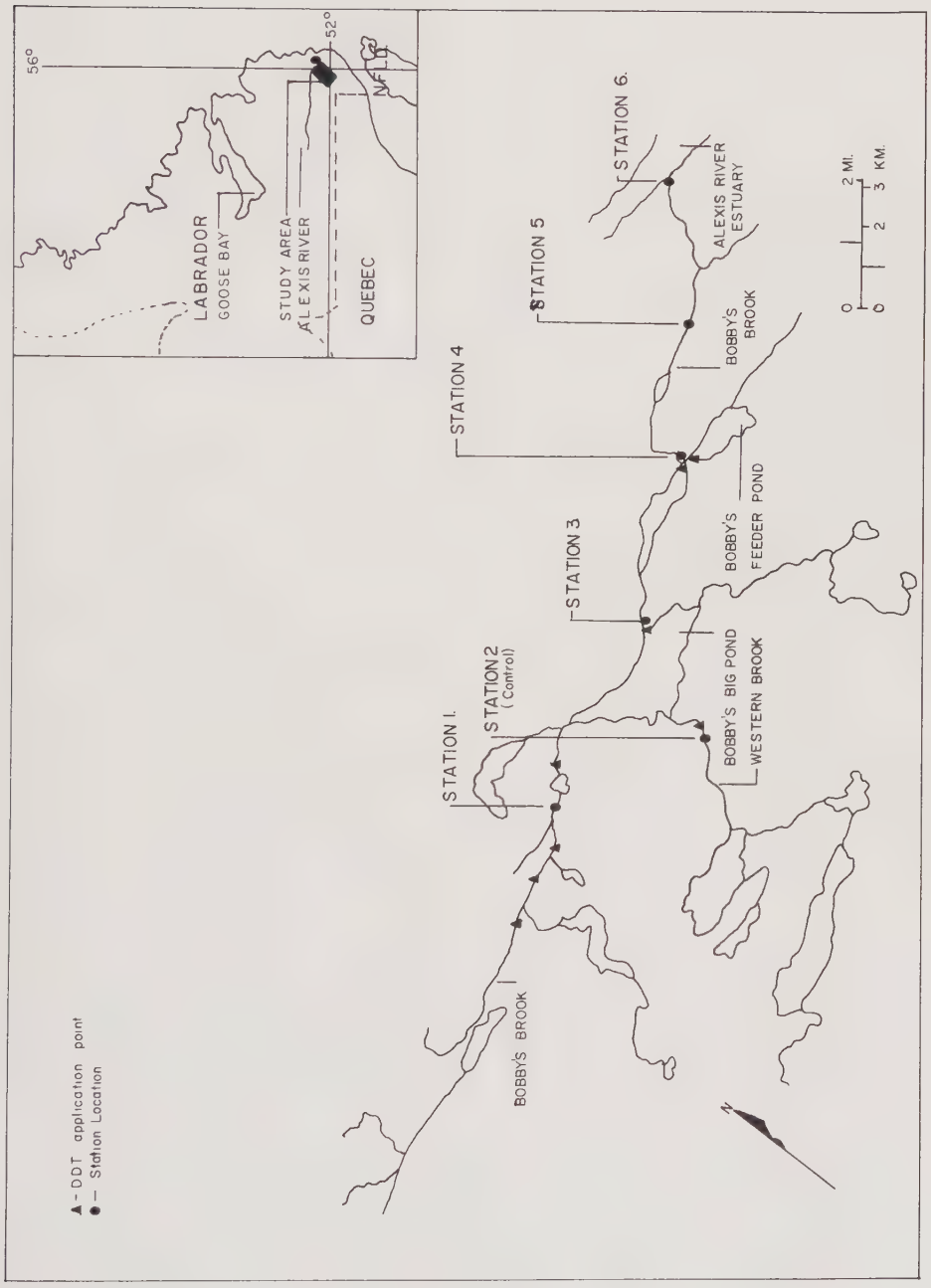

FIGURE 1. *Bobby's Brook sampling stations and DDT application points.*

Results

Bottom Fauna Sampling

Numbers of aquatic insect larvae dropped sharply in the counting plots immediately after each DDT application (Fig. 2). Insect larvae did not decrease in the control plot on the same dates as they did in the experimental plots although numbers did decrease somewhat with time. This was probably due to emergence and disturbance of the larvae. Caddisfly larvae were most affected, but stonefly and mayfly larvae populations also dropped to zero or near zero at all stations except the control (Fig. 3).

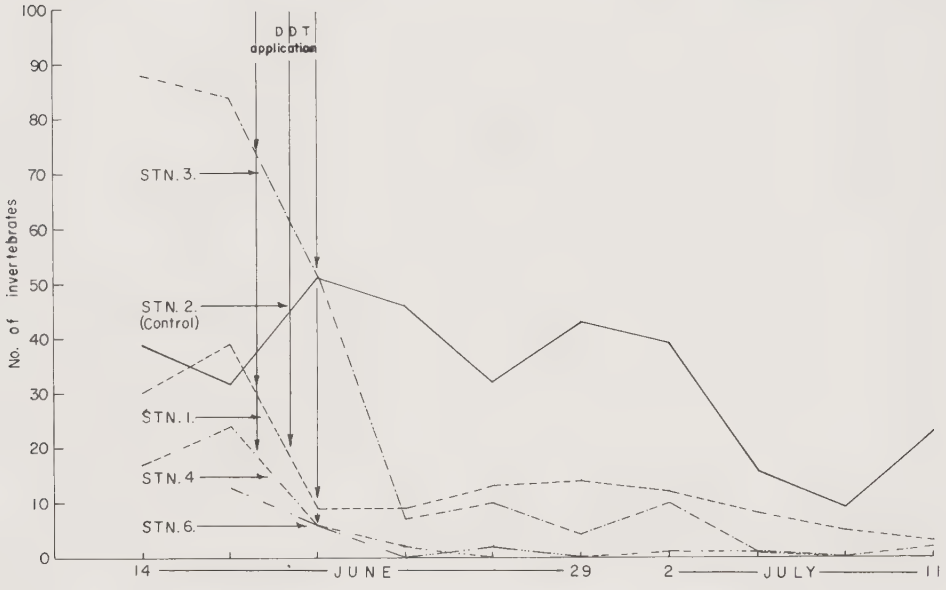

FIGURE 2. *Nos. of bottom invertebrates were reduced by larviciding at all stations except the control.*

Seven days after the reduction in number of bottom insects in the brook, stones previously inhabited by the insects were covered with a new growth of algae. After ten days, a large population of Chironomids inhabited the algae. Black fly larvae were in the counting areas one week after larviciding ended. Invertebrate mortalities were found as far as five miles downstream from the DDT application points.

Drifting Insects

Fifteen minute pre-larviciding net samples always yielded less than ten insect larvae. Larviciding caused an exponential increase in the numbers of drifting insect larvae, totals for the three nets ranging from 45 to 8,000 for a fifteen minute sampling period at the different stations (Fig. 4 and 5).

The immediate effects of DDT on drifting insect larvae lasted over one and one-half hours at Station 3 (Fig. 6).

No black fly larvae or micro invertebrates were collected in the nets because of the mesh size.

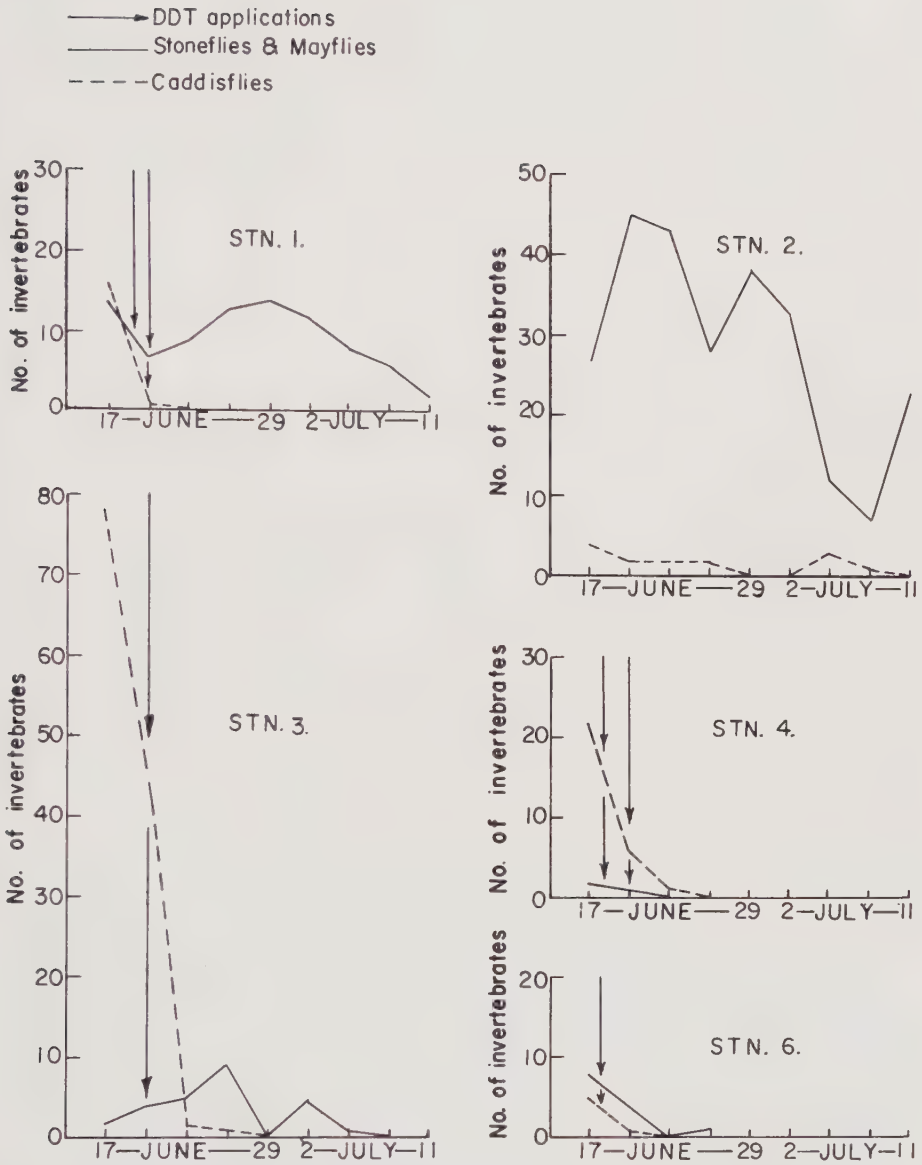

FIGURE 3. *Caddisfly larvae populations were affected more severely than stonefly and mayfly larvae by the DDT larviciding (Note that stonefly and mayfly larvae populations eventually decreased to near zero at all stations except the control).*

Wild Fish

During DDT applications, fish consumed dead insect larvae drifting in the DDT-fuel oil slick. Two days later at *Station 3*, near which two quarts[1] of 10 percent DDT had been applied, 30 dead and 9 dying trout were collected. Eleven dead trout were

[1]The calculated amount for this application point was 2.5 qts., but this was unavailable at the application time.

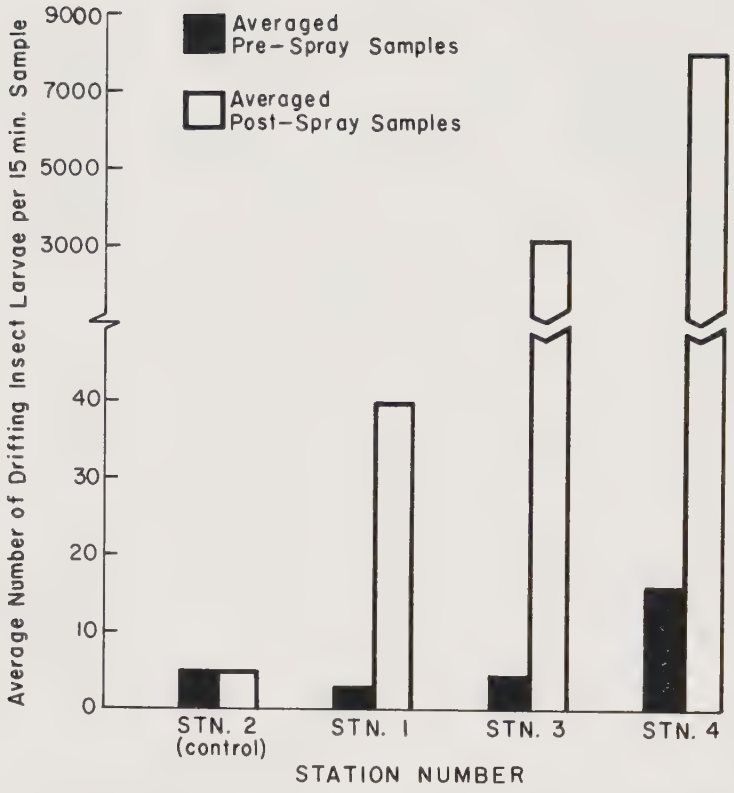

FIGURE 4. *Numbers of drifting insect larvae in the brook before, and after DDT applications. (Note: 90% of those sampled at Stn. 1 were stoneflies; 90% of those sampled at all other stations were caddisflies).*

FIGURE 5. *Drifting insect net after a DDT application; shown in the net are dead caddisfly larvae.*

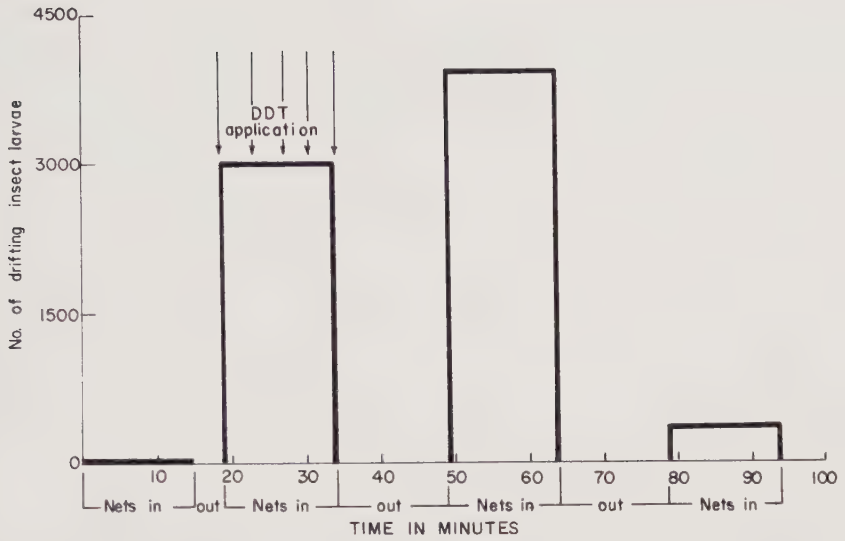

FIGURE 6. *The duration of the immediate effects of DDT on insect larvae inhabiting the bottom of the brook at Stn. 3 was over one and one-half hours. (Note: Width of bar indicates duration of set; first set (0-15 min.) zero count).*

found the same day near Station 4. The dying fish were lying in shallow water and going through nervous convulsions with distention of their mouths and opercles. Some dead fish had a bloated appearance caused by large numbers of caddisfly larvae in their stomachs. Other stomachs of collected dead fish were either half full or completely empty. One fish that had died recently was found at Station 1, 17 days after the last DDT application. The DDT content of its tissue was over ten times that of the controls. Tissue content of fish found dead along the brook ranged from 0.3 to 0.5 ppm as compared to 0–0.03 ppm in control fish[2]. Two killed fish collected along the brook showed stomach content DDT readings of 139.54 and 38.79 ppm. DDD content of these stomach contents (mostly dead insect larvae) was 98.23 and 27.72 ppm.

Higher DDT content readings than the control were found in fish angled at each station four days after larviciding. Live fish taken twenty days post-larviciding contained DDT readings higher than those caught four days port-larviciding (Fig. 7).

Larviciding was halted when it was established to be harmful to fish in the brook.

Netted Wild Fish

Stomachs of forty gill net caught fish taken in Bobby's Big Pond near Station 3 before larviciding contained almost exclusively caddisfly larvae and cases. Stomachs of forty fish netted in the same location twenty days post-larviciding were either entirely empty or contained a few winged insects. The closest DDT application point to the gill net location was three miles upstream.

[2]No explanation for DDT in control fish was found as insecticides had not previously been used within 150 miles of the area. Reed (1966) reported DDT in control fish in Alaska. Woodwell (1967) suggests air and ocean currents are responsible for the distribution of DDT over the earth's surface.

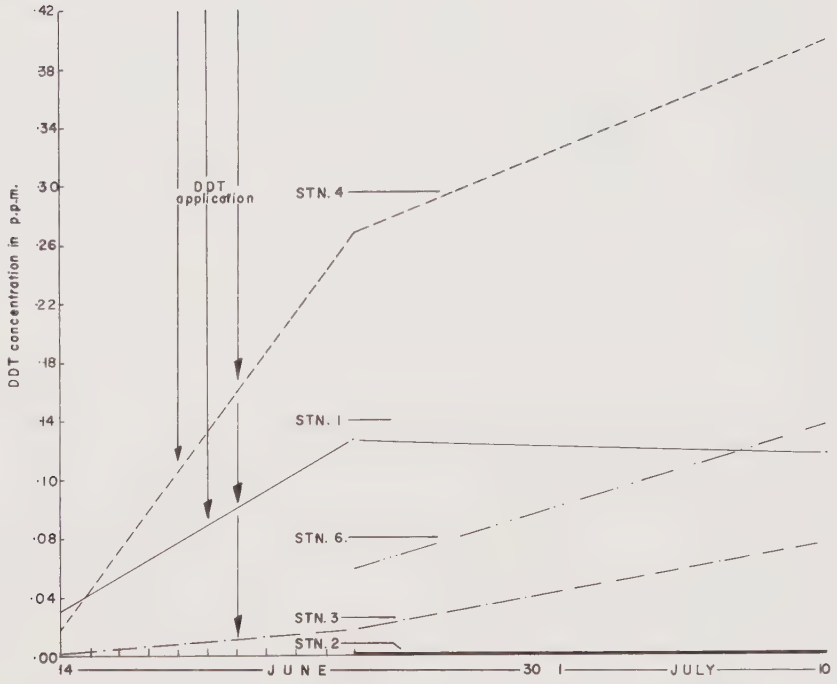

FIGURE 7. *DDT content of wild fish increased with time at the stations sampled. Only trace amounts were found on fish sampled at Stn. 2.*

Caged Fish

Only 3 of the 90 caged fish died during the 29 day experiment. This mortality was statistically insignificant. DDT and metabolite content of the caged fish at the end of the study showed little increase over the control except for a 2.1 ppm value for a fish from Station 3.

Water Analysis

DDT content of the water was highest during the application period, ranging from 0.00379 to 0.01191 ppm. Only trace amounts of DDT were detected one day after larviciding.

General chemistry was similar, pre and post larviciding.

Discussion

There is general agreement in the literature that aerial spraying of DDT over forested regions reduces the bottom fauna of water courses in the spray area (Welch and Spindler, 1964; Corley, Bjorn and Webb, 1965; Ide, 1967; Kerswill, 1967).

Studies by developers of this ground larviciding method led to different conclusions. Gjullin, et al (1949) found 10 ppm DDT over a 15 minute period did not affect caddisfly larvae. Jamnback and Collins (1955), and Twinn and Peterson (1955), state that 0.1 ppm DDTin a stream over the same period did not harm aquatic life other than

black fly larvae. A study on black fly control in the Adirondack Mountains concluded that "there was no profound change in stream fauna after five to ten years of black fly larvae control. . . . " (Jamnback and Eabry, 1962).

In Bobby's Brook, less than 0.1 ppm DDT over a 15 minute period completely destroyed caddisfly populations at the stations sampled. Caddisfly larvae compose from 40–80 percent of the benthic fauna of over 25 major rivers sampled in Labrador (Sollows, et al, MS 1953, 1954; Peet, MS 1966, 1967).

Jenson and Gaufin (1966) found 96 hour TL_m for stoneflies (*Pteronarcys californica* and *Aeroneuna pacifica*) to be 1.8 ppm and 3.2 ppm respectively. Although stonefly mortalities occurred in Bobby's Brook, highest DDT concentration in 36 water samples was 0.011 ppm.

Elson and Kerswill (1966) state that caddisflies took four years to recover, mayflies three years and stoneflies two after DDT spraying in New Brunswick. Ide (1967) found Chironomid populations were fastest in recovering after a DDT kill. He also observed as in Bobby's Brook, a heavy algae growth on the stones after a DDT spray killed insect larvae in a stream (personal communication).

Studies in New Brunswick (Elson and Kerswill, 1966) and Idaho (Corley, Bjorn and Webb, 1965) concluded, as did the Bobby's Brook study, that DDT causes harm to organisms miles downstream of the application point. Hoffman and Droog (1953) found that progressively higher insect mortalities could be directly correlated with higher concentrations of DDT in aquatic vegetation as one sampled downstream along a DDT treated brook.

Numbers of drifting insect larvae increased after a DDT spray in studies by Welch and Spindler (1964) and Reed (1966).

The literature reveals great variability in DDT content of wild fish found dead after spraying or larviciding. Differing lethal values probably vary as individual fish exposure times to DDT and levels of food contamination differ. Contrary to the Bobby's Brook findings, Butler (1965) reported 1.4 ppm DDT in Alaskan fish did not result in their death. Fish mortality on the Bobby's Brook system could always be related to at least a tenfold increase in the DDT content of the fish. As in Labrador, Welch and Spindler (1964) found in Montana that the insecticide content of wild live trout increased with time after a DDT spray.

Elson and Kerswill (1966) found in New Brunswick that wild brook trout populations recovered in two or three years.

The view is held by some that DDT settles out in slow moving water (Dr. A. S. West, personal comm.). No evidence of this was found in the Labrador study. Since the DDT was in an oil solution, it is plausible that it could float across ponds. This seemed even more possible when six mud samples collected for DDT analysis at points where the brook entered Bobby's Big Pond contained no DDT. The long metamorphic cycle of caddisflies makes it unlikely that natural emergence during the twenty days between gill net sampling could account for their complete disappearance from the trout's food spectrum. The abrupt change in diet of pond trout post larviciding indicated DDT affected aquatic life in the pond. Keenleyside (1967) noted a change in the salmon parrs' diet from Diptera, Trichoptera and Ephemeroptera before DDT spraying, to Chironomidae, snails and worms after spraying in New Brunswick.

Caged fish experiments on Bobby's Brook indicated that trout mortality and high DDT content occurred only when they ate insect larvae killed by DDT. This could have been because of the fishs' size (16–24 cm.). DDT killed larvae were in stomachs of dead fish recovered by Welch and Spindler (1964) after a spray. Macdonald (1965), Penney and Macdonald (1966) and Kerswill and Edwards (1967) found kills in caged Atlantic salmon parr in New Brunswick watersheds that were attributed to DDT spraying. Whether the fish died after contact with DDT in water or after accumulating lethal amounts of DDT in their tissue from ingested food was not known.

Adequate sampling of a brook for DDT content is a difficult operation due to the floating oil slick, changing stream eddies and stream depths. The "grab sampling" technique used on Bobby's Brook was not satisfactory. The larviciding method was calculated to give a concentration of 0.1 ppm DDT in the brook. DDT levels in water samples taken from the brook were less than 1/10th this concentration. Considering the biological effects of the DDT on the stream, this seemed unrealistically low. Henderson, Pickering and Tarzwell (1959) found in bioassay experiments that a 96 hour TL_m value of 0.016 ppm DDT was obtained using "a moderately sensitive fish". Katz (1961) reported 96 hour TL_m values of 0.10 ppm or "much less" for salmonids. Abram (1966) found rainbow trout, *Salmo gairdneri* (Richardson, 1836) were killed in less than an hour in water of 0.10 ppm DDT. Allison et al (1964) found threshold levels for cutthroat trout of around 0.10 ppm DDT, for monthly contact with DDT in the water. They also found that 30 mg DDT per kg of fish in the trout's food per week was a threshold level.

It is unlikely the amount of DDT put into the Bobby's Brook System had any effect on Alexis River fauna. If most of the 132 application points had been treated at three week intervals as planned, the Alexis would probably have been affected. The Alexis River is a major producer of Atlantic salmon in Labrador.

Conclusions

1. DDT caused high bottom invertebrate mortality by direct contact.
2. DDT caused mortalities of Eastern brook trout *Salvelinus fontinalis* in the brook by contamination of the fishes' food above their maximum tolerance levels.
3. DDT caused no significant short term fish mortality by direct contact.
4. Larviciding was unsuccessful for black fly control without harm to non-target organisms.

Acknowledgements

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An Improved Dip Net for Use in Electro-Fishing

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Abstract

A new dip net for the capture of small fish, stunned by D. C. electro-fishing, is described. The net is for use in shallow riffles only. Undesirable interference with the electrical field around the anode by the original metal dip net is eliminated by the use of non-conductive materials. The new net is light and safe to use. Detailed drawing and photograph are included.

Introduction

The increased use of direct current electro-fishing for fish population studies in shallow areas of rivers and streams creates the need for a light, efficient, specialized dip net. In the Canadian Maritime Provinces, electro-fishing is used extensively to investigate juvenile populations of Atlantic salmon, speckled trout and other species dwelling in similar habitats. The apparatus used is similar to that described by Smith and Elson (1950). The source of power is a direct-current generator driven by a small air-cooled engine. Voltages from 250 to 650 volts can be produced at very low amperage (0.5-1.0 amps). A rheostat, fuse and other switching devices are mounted in a control box. The positive electrode is shaped like a probe with 250 feet of electrical cable; the cathode is a metal screen (usually copper) placed in contact with the stream bottom in a stationary position. Fine mesh nylon nets are used to enclose the area under study. The fish are collected by immersing the positive electrode in the water and gathering narcotized fish with special dip nets.

The original dip nets consisted of a flattened steel hoop with special lip or blade for use near the stream bottom. The hoop was attached to a galvanized iron pipe handle insulated with plastic tape. While this arrangement had the desired stiffness, the metal components interferred with the electrical field around the anode probe. The lines of force of the electrical field around the probe are broken when an electrical conductor is placed between fish in electro-narcosis and the electrical source. The fish recover immediately and escape. This effect was demonstrated by setting up a small experiment using a two-compartment live box. The bottom, the two sides and the partition were made of plywood. The ends were screened, one side with ordinary copper fly screen, the other with nylon fly screen. Fish were placed in both compartments and the probe was placed within normal effective distance. As the current was turned on near the nylon-screened compartment, all the fish were immediately stunned. When the probe, with current turned on, was placed near the copper-screened compartment of the live box, the fish were unaffected.

The quantitative method for population density studies, using electro-fishing apparatus, calls for a consistently large fishing effort. To increase efficiency, a dip net was assembled using non-conductive materials which would in no way interfere with the lines of force of the electrical field around the anode.

Description of the Dip Net

Materials

The basic components of the modified dip net are: a three-foot length of half-inch, commercial-grade nylon rod bent at 400°F. into a triangular shape with rounded

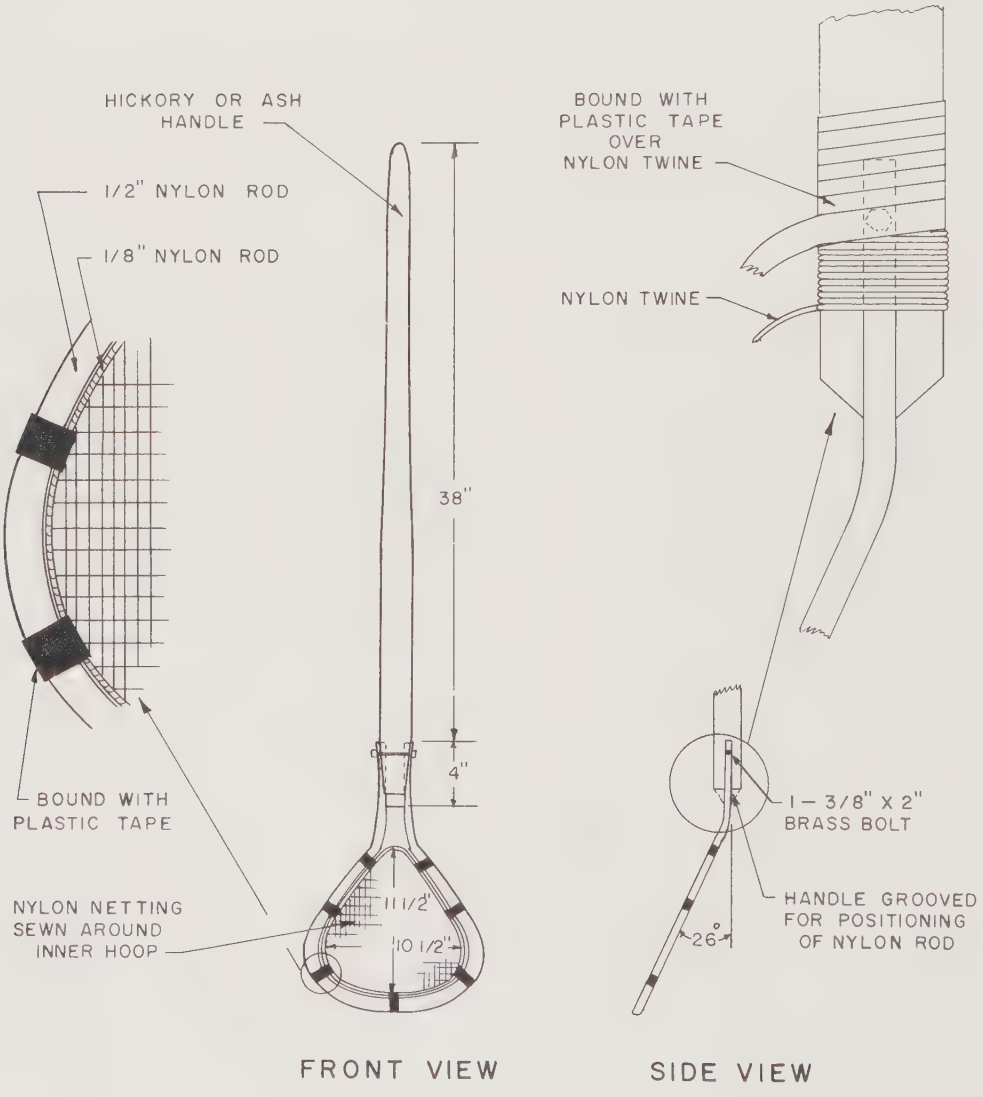

FIGURE 1. *Details of dip net construction.*

corners; a four or five foot-long hickory or ash handle of the type used for garden tools; a few square feet of 100% nylon netting; and some light nylon twine and electrical plastic tape.

Assembly

A groove is prepared on each side of the hickory-wood handle to receive the two mounting ends of the nylon frame. A ⅜″ cadmium-plated or solid brass bolt is pushed through a hole drilled through the wooden handle and both ends of the nylon rod. The bolt is the only metal part used in making the dip net. The mounting ends are then lashed firmly to the hickory handle with a light nylon twine or with waxed cotton twine (whipping twine), and the mounting section is taped with black plastic tape to insulate the metal bolt and protect the whipping. The dip net frame is attached to the handle at an angle of approximately 26 degrees (Figure 1). This angle, however,

FIGURE 2. *The improved dip net.*

can vary to suit the operator. The netting is sewn, with a little sag, to a sub-frame made of ⅛″ nylon rod which in turn is secured at several points to the main frame with plastic tape. (Figure 1 and 2). This protects the netting from chafing against the stream bottom.

Field Assessment

Qualitative observations on the efficiency of the new dip net were made during the field seasons of 1966 and 1967 (in 1967 three of these nets were in use at different locations in the Maritime Provinces). It was found that the new dip net was rigid enough to cope with the water velocities encountered in shallow riffle areas. It could be swept through fast water without buckling backwards, and did not interfere with attraction of fish to the probe.

The modified net also has the advantage of being safer for the operator. The original type of dip net had an insulated steel handle which was subject to chafing during transport over rough terrain, and has caused shocks to users.

The use of the modified dip net is most effective in the capture of fish to a maximum length of 12 centimeters. Its light weight and non-conductive qualities make it a safe, more efficient tool, and it can easily be assembled in a few hours for a few dollars worth of materials.

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